AQUATIC EFFECTS TECHNOLOGY EVALUATION (AETE) PROGRAM

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Final Report
Mattabi Mine Site,
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1997 FIELD PROGRAM - AETE MATTABI MINE

SITE REPORT

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1997 FIELD PROGRAM - AETE
MATTABI MINE

SITE REPORT

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AQUATIC EFFECTS TECHNOLOGY EVALUATION PROGRAM

Notice to Readers

1997 AETE Field Program
Mattabi Mine Site, Ontario

The Aquatic Effects Technology Evaluation (AETE) program was established to review appropriate technologies for assessing the impacts of mine effluents on the aquatic environment. AETE is a cooperative program between the Canadian mining industry, several federal government departments and a number of provincial governments; it is coordinated by the Canada Centre for Mineral and Energy Technology (CANMET). The program was designed to be of direct benefit to the industry, and to government. Through technical and field evaluations, it identified cost-effective technologies to meet environmental monitoring requirements. The program included three main areas: acute and sublethal toxicity testing, biological monitoring in receiving waters, and water and sediment monitoring.

The technical evaluations are conducted to document certain tools selected by AETE members, and to provide the rationale for doing a field evaluation of the tools or provide specific guidance on field application of a method. In some cases, the technical evaluations included a go/no go recommendation that AETE takes into consideration before a field evaluation of a given method is conducted.

The technical evaluations are published although they do not necessarily reflect the views of the participants in the AETE Program. The technical evaluations should be considered as working documents rather than comprehensive literature reviews. The purpose of the technical evaluations was to document specific monitoring tools. AETE committee members would like to stress that no one single tool can provide all the information required for a full understanding of environmental effects in the aquatic environment.

For more information on the monitoring techniques, the results from their field application and the final recommendations from the program, please consult the AETE Synthesis Report to be published in the spring of 1999.

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EXECUTIVE SUMMARY

The Mattabi Mine (Ontario) study is one of four field evaluations carried out in 1997 under the Aquatic Effects Technology Evaluation (AETE) Program, a joint government-industry program to evaluate the cost-effectiveness of technologies for the assessment of mining-related impacts in the aquatic environment. The other three mines studied were Dome (Ontario), Myra Falls (British Columbia) and Heath Steele (New Brunswick). Results of all four studies are summarized and evaluated in a separate summary report.

Mattabi Mines of Noranda Inc. operated a copper-lead-zinc open pit and underground mining operation and concentrator complex 80 km northeast of Ignace in northwestern Ontario. The mine was closed in 1991 and is undergoing rehabilitation; however, site runoff and seepages continue to be collected, treated and discharged to control impacts from acid rock drainage (ARD) and metal leaching. Effluent is treated with lime, polished and discharged to Bell Creek, which flows into Sturgeon Lake. Sturgeon Lake and other smaller lakes (including No Name Lake) were historically affected by ARD and loadings of zinc and other metals, and sediments in these waters are enriched with metals.

The objectives of the 1997 field program were to test 13 hypotheses formulated under four guiding questions:

1. are contaminants getting into the system (and to what degree and in which compartments)?
2. are contaminants bioavailable?
3. is there a measurable (biological) response? and
4. are contaminants causing the responses?

The hypotheses are more specific questions about the ability or relative ability of different monitoring tools to answer these four general questions about mine effect. The evaluation of tools included: sediment monitoring (sediment toxicity tests); fish monitoring (tissue metallothionein and metal analyses, and population/community indicators), and; integration of tools (relationships between exposure and biological responses and use of effluent sublethal toxicity).

Of the 13 hypotheses, 12 were tested at Mattabi as outlined in Table 1.1. The remaining hypothesis not tested at Mattabi relates to effluent chronic toxicity and its linkage to benthic and fish effects (H13). This hypothesis was deleted to optimize the field work plan (for project budget).

The sediment quality triad was used as an additional means of evaluating the linkages between sediment toxicity, sediment chemistry and benthic community response (H10 and H11) in a sediment quality gradient in No Name Lake and Mine Creek Bay (Sturgeon Lake). The triad provides a more holistic means of evaluating the tools.

Study Design

The study design at Mattabi was based on lake sampling for benthos, sediment chemistry and sediment toxicity using a gradient design, including five exposure areas having different sediment

zinc concentrations (No Name Lake, Mine Creek Bay) and two reference areas (Tag Lake and Peterson Cove-Sturgeon Lake). Three stations were sampled for sediment quality, toxicity and benthos within each of the seven areas.

Sampling in Bell Creek followed a reference-exposure (Control-Impact) design, and allowed for testing of fish-related hypotheses. The reference area for fish collection was located in generally similar habitat nearby in a separate watershed (English River). Four stations were sampled for fish in each of the two areas, providing replicates for testing of hypotheses relating to fish populations and communities (H5 and H6).

Sampling Program

The field survey at Mattabi was completed in mid-October 1997, and included:

- water sampling at each of seven sediment/benthic sampling areas (one each) and each of two fish sampling areas (four per area). Only Bell Creek and English River data were used in hypothesis testing;

- surficial sediment sampling at each of 21 sampling stations (7 areas x 3 stations) using a petite Ponar, for determination of "total" metal concentration, partial metal concentration (i.e., the Fe and Mn oxide-bound fraction) and concentrations of acid volatile sulphide (AVS) and simultaneously extracted metals (SEM);

- surficial sediment sampling at the same 21 stations for benthic macroinvertebrate community analysis and for sediment toxicity testing (*Hyalella azteca* - survival and growth, *Chironomus riparius* - survival and growth, *Tubifex tubifex* - survival and growth);

- sampling of the fish community by gillnet (at least two sets per station) at each of four exposure stations and four reference stations for determination of fish community and population parameters, and to provide specimens of white sucker and northern pike for tissue analysis; and

- sampling of 83 white sucker and 85 northern pike total from the two sampling areas for analysis of metals and metallothionein (MT) in muscle (metals only), liver, kidney and gill.

Data Overview

Water Quality

Mattabi effluent was enriched in zinc, calcium and sulphate. Bell Creek water contained elevated concentrations of zinc, copper, cadmium and lead relative to the English River, with zinc concentrations greater than Canadian Water Quality Guidelines downstream of the mine.

Concentrations of zinc were also elevated in water in No Name Lake and to a much lesser degree in Mine Creek Bay relative to the lake reference areas. Water quality guidelines were exceeded

only in No Name Lake, with zinc concentration nearly two orders of magnitude above the guideline.

Dissolved and total metal concentrations showed similar spatial patterns for key metals (Zn, Cu, Cd, Pb), with dissolved and total values similar in magnitude in most cases except for Pb in No Name Lake (dissolved Pb < total Pb).

Sediment Quality

All sediments were organic-rich and sandy, with total organic carbon contents of 20 to 30%. Concentrations of total Zn, Pb, Cd, Cu, As, Ni and Hg exceeded Canadian Interim Sediment Quality Assessment Values at most exposure stations, with maximum values for zinc as high as two orders of magnitude above Probable Effect Levels (PELs). Metal concentrations followed a declining concentration gradient across the range of exposure areas sampled.

Partial metal concentration gradients were weaker than those for total metals, implying that the largest fraction of metals present is relatively strongly bound to sediment particles.

The SEM/AVS ratios showed no distinct spatial patterns, although SEM and AVS values were much greater in the exposure gradient than at reference areas. The results imply some potential for sediment toxicity, although ratios were relatively low (≤1.5) in most exposure areas (except No Name Lake where higher ratios occurred). A ratio above one implies some potential for sediment toxicity, especially when SEM values are high.

Sediment Toxicity

Sediment toxicity (lethal and sublethal effects) showed little or no response in any test species to the sediment quality gradient at Mattabi. Although mortality occurred both in *Chironomus* and *Hyalella*, responses appeared similar regardless of location (reference vs exposure).

Benthic Macroinvertebrates

Benthic community trends were apparent in the sediment quality gradient, with reduced numbers of taxa in the most effected sediments. Hydracarina (water mites), *Caenis* (mayfly) and *Pisidium* (pea clam) in particular were reduced in abundance at the most affected locations.

Fish

Catches-per-unit-effort for all fish species combined were not distinctly different between reference and exposure areas. White sucker appeared more abundant in the exposure area and northern pike more abundant in the reference area. In white sucker, growth, liver size, gonad size and fecundity were comparable in reference and exposure fish. In pike, fish size at age, organ sizes and fecundity were distinctly greater in exposed (Bell Creek) fish relative to the reference area. When adjusted for body weight, organ sizes remained greater in exposed pike.

Tissue metal concentrations showed some reference-exposure differences, with differences most pronounced for selenium in all tissues and both species. This could be mine-related as selenium was detected in Mattabi effluent, but not in any other water sample.

The only metallothionein response measured was in gill and kidney of northern pike. No metallothionein responses occurred in sucker.

Hypothesis Testing

Hypothesis testing results are summarized in Table 5.2. Results of testing indicate that some of the contaminants (metals) are bioavailable, that some biological responses occurred and that contaminants appear to cause some of these responses.

Technology Evaluation

Some of the tools evaluated demonstrated a mine effect at Mattabi, whereas others did not (Table 6.2). Monitoring tools demonstrating effects or partially demonstrating effects included most water and sediment chemistry tools (except SEM/AVS), most benthic macroinvertebrate tools, and some of the fish tissue and metallothionein tools (depending on metal, tissue type and fish species). Tools that did not show responses consistent with impact included several in the fish health, fish population/community and sediment toxicity tool boxes. The limited effectiveness of some of these tools may be due to low metal bioavailability, possibly combined with the effects of natural variation. Of the monitoring tools in the same "tool box" demonstrating effects (e.g., total vs partial metals in sediments; metals vs metallothionein in tissues, etc.), major differences in tool effectiveness were not apparent at Mattabi (Table 6.3).

Overall, the relatively subdued impacts of metals at Mattabi are unexpected and noteworthy. This general condition contrasts with the greater bioavailability of metals and impact at Heath Steele and Myra Falls. This is particularly unusual because metal concentrations in Mattabi sediments are greater than observed at any of the other mines studied in the AETE program.

Conclusions on the cost-effectiveness of the tools based on results from all four mine sites studied in 1997 are found in a separate document "Summary and Cost-Effectiveness Evaluation of Aquatic Effects Monitoring Technologies Applied in the 1997 AETE Field Evaluation Program".

SOMMAIRE

L'étude du site de la mine Mattabi (Ontario) est l'une des quatre évaluations sur le terrain effectuées en 1997 dans le cadre du Programme d'évaluation des techniques de mesure d'impacts en milieu aquatique (ETIMA), programme conjoint gouvernement-industrie destiné à évaluer le rapport coût-efficacité des technologies d'évaluation des impacts liés aux activités minières dans le milieu aquatique. Les trois autres sites miniers étudiés étaient ceux de Dome (Ontario), de Myra Falls (Colombie-Britannique) et de Heath Steele (Nouveau-Brunswick). On présente un résumé et une évaluation des résultats de ces quatre études dans un rapport sommaire distinct.

La mine Mattabi de Noranda Inc. exploitait une mine à ciel ouvert de cuivre, de plomb et de zinc combinée à une mine souterraine et à une installation de concentration à 80 km au nord-est d'Ignace, dans le nord-ouest de l'Ontario. Cette mine a été fermée en 1991 et on y effectue actuellement des travaux de restauration; toutefois, on continue à recueillir des eaux de ruissellement et d'infiltration, qui sont traitées avant leur rejet afin de limiter l'impact des eaux d'exhaure des roches acides et de la lixiviation des métaux. On traite les effluents avec de la chaux, on les purifie et on les rejette dans le ruisseau Bell, qui se déverse dans le lac Sturgeon. Au cours des années passées, on a noté dans ce lac et dans d'autres lacs plus petits (notamment le lac No Name) des effets d'eaux d'exhaure acides, de charges de zinc et d'autres métaux, ainsi que de sédiments enrichis en métaux.

Les objectifs du programme sur le terrain de 1997 étaient de vérifier 13 hypothèses formulées pour tenter de répondre à quatre questions principales :

1. Est-ce que les contaminants pénètrent dans le réseau aquatique (et dans l'affirmative, dans quelle mesure et dans quels compartiments)?
2. Les contaminants sont-ils biodisponibles?
3. La réponse (biologique) est-elle mesurable?
4. Les contaminants sont-ils la cause de ces réponses?

Ces hypothèses représentent des questions plus spécifiques concernant la capacité (relative) des différents outils de surveillance de répondre à ces quatre questions générales sur les effets des activités minières. L'évaluation des outils prévoyait notamment la surveillance des sédiments (tests de toxicité des sédiments), la surveillance des poissons (dosage de la métallothionéine et des métaux dans les tissus et détermination des indicateurs des populations/communautés) et, enfin, l'intégration des outils (rapports entre l'exposition et les réponses biologiques, et utilisation de la toxicité sublétale des effluents).

On a vérifié 12 des 13 hypothèses au site de la mine Mattabi (voir le tableau 1.1). Les hypothèses non vérifiées à ce site concernent la toxicité chronique des effluents et ses liens avec les effets benthiques et les effets sur les poissons (H13). On a retiré cette hypothèse de la liste afin d'optimiser le plan de travail sur le terrain (pour le budget du projet).

On a utilisé les trois paramètres de la qualité des sédiments comme outil supplémentaire pour l'évaluation des liens entre la toxicité des sédiments, la chimie des sédiments et la réponse de la communauté benthique (H10 et H11) dans un gradient de qualité des sédiments du lac No

Name et de la baie Mine Creek (lac Sturgeon). Ces trois paramètres donnent une vue plus générale pour l'évaluation des outils.

Plan de l'étude

Le plan de l'étude à la mine de Mattabi était basé sur l'échantillonnage des lacs pour l'étude du benthos, ainsi que de la chimie et de la toxicité des sédiments selon un gradient, et notamment dans cinq zones d'exposition dont les sédiments présentaient différentes concentrations de zinc (lac No Name, baie Mine Creek) et dans deux zones de référence (lac Tag et anse Peterson - lac Sturgeon). On a prélevé des échantillons dans trois stations de chacune de ces sept zones pour déterminer la qualité et la toxicité des sédiments, ainsi que pour l'étude du benthos.

Dans le ruisseau Bell, l'échantillonnage suivait un modèle zone de référence (témoin) - zone d'exposition (impact), qui permettait de vérifier des hypothèses liées aux poissons. La zone de référence pour le collecte des poissons était située dans un habitat à peu près semblable dans un bassin hydrographique voisin, mais distinct (rivière English). On a prélevé des échantillons de poissons dans quatre stations de chacune des deux zones, de façon à obtenir des exemplaires multiples pour vérifier les hypothèses concernant les populations et les communautés de poissons (H5 et H6).

Programme d'échantillonnage

On a terminé les relevés sur le terrain à Mattabi vers la mi-octobre 1997, et notamment :

- l'échantillonnage de l'eau dans chacune des sept zones d'échantillonnage pour les sédiments ou les organismes benthiques (un pour chaque zone) et dans chacune des deux zones d'échantillonnage pour les poissons (quatre par zone). On n'a utilisé que les données du ruisseau Bell et de la rivière English pour la vérification des hypothèses;

- l'échantillonnage des sédiments de surface à chacune des 21 stations d'échantillonnage (7 zones x 3 stations) à l'aide d'un échantillonneur « Petite Ponar » pour la détermination de la concentration « totale » des métaux, de la concentration partielle des métaux (c.-à-d. la fraction liée aux oxydes de Fe et de Mn), ainsi que des concentrations de sulfure volatil en milieu acide et des métaux extractibles simultanément;

- l'échantillonnage des sédiments de surface à ces 21 stations pour l'analyse de la communauté des macroinvertébrés benthiques et pour des tests de toxicité des sédiments (survie et croissance d'*Hyalella azteca*, de *Chironomus riparius* et de *Tubifex tubifex*);

- l'échantillonnage des communautés de poissons à l'aide de filets maillants (au moins deux échantillons par station) à chacune des quatre stations d'exposition et à quatre stations de référence pour la détermination des paramètres des communautés et des populations de poissons, ainsi que pour obtenir des spécimens de meunier noir et de grand brochet pour des analyses de tissus;

- l'échantillonnage d'un nombre total de 83 meuniers noirs et de 85 grands brochets de deux zones d'échantillonnage pour le dosage des métaux et de la métallothionéine (MT) des muscles (métaux seulement), du foie, des reins et des branchies.

Aperçu des données

Qualité de l'eau

Les effluents du Mattabi étaient enrichis en zinc, en calcium et en sulfate. L'eau du ruisseau Bell contenait des fortes concentrations de zinc, de cuivre, de cadmium et de plomb par rapport à celles de la rivière English, ainsi que des concentrations de zinc dépassant les limites des Recommandations pour la qualité des eaux au Canada en aval de la mine.

De plus, les concentrations de zinc étaient élevées dans l'eau du lac No Name, ainsi que dans la baie Mine Creek (mais beaucoup moins dans celle-ci) par rapport aux zones de référence du lac. Les limites des Recommandations n'étaient dépassées que dans les eaux du lac No Name, où l'on observait un dépassement de presque deux ordres de grandeur pour le zinc.

Pour les concentrations de principaux métaux dissous et totaux (Zn, Cu, Cd, Pb), on observait, dans la plupart des cas, des profils de distribution spatiale semblables, avec des valeurs du même ordre pour les métaux dissous et totaux, sauf pour le Pb du lac No Name (Pb dissous plus faible que le Pb total).

Qualité des sédiments

Tous les sédiments étaient riches en matières organiques et sableux, avec des teneurs en carbone organique total de 20 à 30 %. Les concentrations de Zn, Pb, Cd, Cu, As, Ni et Hg totaux dépassaient les valeurs de l'évaluation intérimaire canadienne de la qualité des sédiments (Canadian Interim Sediment Quality Assessment Values) à la plupart des stations d'exposition, avec des valeurs maximales pour le zinc dépassant de jusqu'à deux ordres de grandeur les teneurs à effets probables. Les concentrations des métaux suivaient un gradient de concentration décroissant sur la plage des valeurs observées dans les zones d'exposition échantillonnées.

Les gradients des concentrations partielles des métaux étaient plus faibles que ceux des concentrations totales des métaux, ce qui indique que la fraction la plus importante des métaux est liée de façon relativement forte aux particules de sédiments.

Les rapports des concentrations des sulfures volatils en milieu acide et de celles des métaux extractibles simultanément ne correspondaient à aucun profil distinct de distribution spatiale, même si dans le gradient d'exposition, les valeurs de ces concentrations étaient très supérieures à celles observées dans les zones de référence. Ces résultats indiquent une certaine possibilité de toxicité des sédiments, bien que les valeurs de ces rapports soient relativement faibles (inférieures ou égales à 1,5) dans la plupart des zones d'exposition (sauf dans le lac No Name, où l'on a observé des valeurs de rapports plus élevées). Une valeur de rapport supérieure à l'unité indique une certaine possibilité de toxicité des sédiments, surtout avec des valeurs élevées de métaux extractibles simultanément.

Toxicité des sédiments

Au site Mattabi, la toxicité des sédiments (effets létaux et sublétaux) correspondait à une réponse faible à nulle du gradient de qualité des sédiments pour toutes les espèces testées. Malgré la mortalité observée chez *Chironomus* et *Hyalella*, les réponses paraissaient semblables quel que soit l'emplacement (comparaison entre la zone de référence et la zone d'exposition).

Macroinvertébrés benthiques

On notait certaines tendances des communautés benthiques dans le gradient de qualité des sédiments, avec des nombres réduits de taxons dans la plupart des sédiments touchés. À ces endroits, on observait notamment des réductions d'abondance chez *Hydracarina* (hydrachnidés), *Caenis* (éphémères) et *Pisidium* (pisidies).

Poissons

Pour toutes les espèces de poissons combinées, on n'observait pas de différences marquées entre les prises par unité d'effort de la zone de référence et celles de la zone d'exposition. On notait une plus grande abondance apparente de meunier noir dans la zone d'exposition, et de grand brochet dans la zone de référence. Pour le meunier noir, la croissance, la taille du foie et des gonades et la fécondité étaient comparables chez les poissons de la zone de référence et de la zone d'exposition. Pour le brochet, les valeurs de taille des poissons selon l'âge, ainsi que celles de la taille des organes et de la fécondité, étaient beaucoup plus grandes chez les poissons exposés (ruisseau Bell) que chez ceux de la zone de référence. Après un ajustement pour tenir compte du poids corporel, les valeurs de taille des organes restaient plus élevées chez les brochets exposés.

Les concentrations de métaux dans les tissus mesurées dans la zone de référence présentaient certaines différences par rapport à celles de la zone d'exposition, et ces différences étaient le plus prononcées pour le sélénium dans tous les tissus chez ces deux espèces. Cet effet pourrait être lié aux activités minières, étant donné qu'on a détecté la présence de sélénium dans les effluents de la mine Mattabi, mais dans aucun autre échantillon d'eau.

On n'a noté une réponse de la métallothionéine que pour les branchies et les reins du grand brochet, et non pour le meunier noir.

Vérification des hypothèses

Les résultats des vérifications des hypothèses sont résumés au tableau 5.2; ils indiquent que certains contaminants (métaux) sont biodisponibles, qu'on observe certaines réponses biologiques et que les contaminants peuvent être à l'origine de certaines de ces réponses.

Évaluation des techniques

Avec certains des outils évalués, on notait un effet dû aux activités de la mine Mattabi, ce qui n'était pas le cas pour d'autres (tableau 6.2). Les outils de surveillance indiquant des effets, même partiels, étaient notamment la plupart des outils de chimie de l'eau et des sédiments (sauf pour le rapport des concentrations des sulfures volatils en milieu acide et de celles des métaux

extractibles simultanément), la plupart des outils d'étude des macroinvertébrés benthiques et certains outils d'analyse des tissus des poissons et de la réponse de métallothionéine (selon le métal, le type de tissu et l'espèce de poisson). Les outils avec lesquels on n'obtenait pas de réponse cohérente en fonction de l'impact étaient notamment plusieurs trousses d'outils pour la détermination de la santé des poissons, des paramètres des populations ou des communautés de poissons et de la toxicité des sédiments. L'efficacité limitée de certains de ces outils pourrait être due à une faible biodisponibilité des métaux, peut-être combinée aux effets des variation naturelles. Au site Mattabi, on n'a pas observé de différences importantes concernant l'efficacité des outils de surveillance de la même « trousse d'outils », avec lesquels on notait des effets (p. ex. concentrations totales par rapport aux concentrations partielles de métaux dans les sédiments; métaux par rapport à la métallothionéine dans les tissus, etc.) (tableau 6.3).

Dans l'ensemble, les impacts relativement modestes des métaux de la mine Mattabi sont inattendus et remarquables, car les conditions générales sont très différentes des conditions de plus grande biodisponibilité des métaux et d'impact aux sites Heath Steele et Myra Falls. Cela est d'autant plus difficile à expliquer que les concentrations de métaux dans les sédiments du site Mattabi sont plus élevées que celles observées à tous les autres sites miniers étudiés dans le cadre du programme ETIMA.

Un document distinct, « Summary and Cost-Effectiveness Evaluation of Aquatic Effects Monitoring Technologies Applied in the 1997 AETE Evaluation Program », présente les conclusions sur le rapport coût-efficacité de ces outils, qui sont basées sur les résultats obtenus pour les quatre sites miniers étudiés en 1997.

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- Effluent Toxicity Test Reports : Myra Falls, Placer Dome, Heath Steele
- Water Sample Collection Methods Applied in the 1997 AETE Field Evaluations
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- Water Chemistry Reports : Myra Falls, Placer Dome, Heath Steele, Mattabi
- AVS/SEM Sediment Chemistry Reports : Myra Falls, Placer Dome, Heath Steele
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- Heath Steele Detailed Periphyton Results – Species and Biomass Chemistry Data
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ANNEX 2: Additional Tool Evaluations
(available upon request from CANMET, Natural Resources Canada)

1.0 INTRODUCTION

The Assessment of the Aquatic Effects of Mining in Canada (AQUAMIN), initiated in 1993, evaluated the effectiveness of Canada's *Metal Mining Liquid Effluent Regulations* (MMLER). One of the key recommendations of the 1996 AQUAMIN Final Report is that a revised MMLER include a requirement that metal mines conduct Environmental Effects Monitoring (EEM), to evaluate the effects of mining activity on the aquatic environment, including fish, fish habitat and the use of fisheries resources.

In parallel, the Canada Centre for Mineral and Energy Technology (CANMET) is coordinating a cooperative government-industry program, the Aquatic Effects Technology Evaluation (AETE) program, to review and evaluate technologies for the assessment of mining-related impacts in the aquatic environment. The intention of the AETE program is to evaluate and identify cost-effective technologies to meet environmental monitoring requirements at mines in Canada. The program is focused on evaluation of environmental monitoring tools that may be used for a national mining EEM program, baseline assessments or general impact studies.

The three principal components of the AETE program are lethal and sublethal toxicity testing of water/effluents and sediments, biological monitoring in receiving waters, and water and sediment chemistry assessments. The program includes both literature-based technical evaluations and comparative field programs at candidate sites. The AETE program is presently at the stage of evaluating selected monitoring methods at four case study sites across Canada.

An AETE Pilot Field Study was carried out in the Val d'Or region of Quebec in 1995 to evaluate a large number of environmental monitoring methods and to reduce the list of monitoring technologies for further evaluation at a cross-section of mine sites across Canada (BEAK, 1996). In 1996, a field evaluation program was initiated and involved preliminary sampling at seven candidate mine sites with the objective of identifying a short-list of mines that had suitable conditions for further detailed monitoring and testing of hypotheses relating to the AETE program. Preliminary study designs were developed for four sites that were deemed to be most suitable for hypotheses testing in 1997 (EVS *et al.*, 1997). The sites selected were Heath Steele, New Brunswick; Lupin, Northwest Territories; Dome Mine, Ontario; and Westmin Resources (now Boliden-Westmin), British Columbia. Lupin was subsequently dropped based on a 1997 reconnaissance survey and replaced with the Mattabi Mines Ltd. site in northwestern Ontario

(BEAK and GOLDER, 1998a). This report documents the results of the 1997 Field Evaluation at the Mattabi Mine site.

The 1996 Field Evaluation constituted Phase I of the Field Evaluation Program. The 1997 Field Evaluation consists of Phases II and III of the Program. Phase II includes the review of necessary background information, finalization of a study design and implementation of the field studies. Phase III includes the compilation, interpretation and reporting of results. Mattabi was not studied in Phase II, but sufficient background information was available to quickly develop and finalize a study design for this site as an alternate to the Lupin Mine.

1.1 Study Objectives

The overall goal of the AETE Program is to identify cost-effective methods and technologies that are suitable for assessing aquatic environmental effects caused by mining activity. An effect is defined as "a measurable difference in an environmental variable (chemical, physical or biological) between a point downstream (or exposed to mining) in the receiving environment and an adequate reference point (either spatial or temporal)". Based on this definition, the AETE Technical Committee developed a series of hypotheses to be tested under field conditions at a number of mine sites in Canada. The Committee agreed that specific hypotheses should be articulated in order to clarify the purpose of the program elements. For the formulation of the hypotheses, the definition of an effect was refined by the AETE Committee to distinguish between effects or responses as measured in biological variables as opposed to effects reflected in physical or chemical changes.

The questions used in developing the hypotheses to be tested in the 1997 Field Evaluation Program were:

1. Are contaminants getting into the system (and to what degree, and in which compartments)? This question relates to the presence of elevated concentrations of metals in environmental media (e.g., water, sediments), and requires an understanding of metal dispersal mechanisms, chemical reactions in sediment and water, and aquatic habitat features which influence exposure of biological communities.

2. Are contaminants bioavailable? This question relates to the presence of metals in biota or to indicators of bioaccumulation such as the induction of metallothionein (MT) in fish. Only if contaminants are bioavailable can a biological effect from chemical contaminants occur.

3. Is there a measurable response? Biological responses may occur only if contaminants are entering the environment and occur in bioavailable forms. These responses may occur at various levels of biological organization, including sub-organism levels (e.g., histopathological effects), at the organism level (e.g., as measured in toxicity testing), or at population and community levels (as measured in resident benthos and fish communities).

4. Are contaminants causing the responses? This question is difficult to measure in field studies directly, as cause-effect mechanisms are difficult to assess under variable conditions prevailing in nature. However, correlations between measures of exposure, chemical bioavailability and response may be used to develop evidence useful in evaluating this question.

The AETE Technical Committee developed a study framework, using the above questions and the three components (water and sediment monitoring, biological monitoring in receiving waters and toxicity testing). The following eight areas of work were identified to finalize the work plan, develop the hypotheses, prioritize issues and identify field work requirements:

1. Chemical presence;
2. The overlap between communities and chemistry testing to determine whether biological responses are related to a chemical presence (bioavailability of contaminants);
3. Biological response in the laboratory;
4. Biological response in the field;
5. Chemical characteristics of the water and sediments used to predict biological responses in the field (contaminants causing a response);
6. The overlap between biological communities responses and bioassay responses to evaluate whether wild community changes are predicted by bioassay responses;
7. The overlap between chemistry and bioassay responses to evaluate whether chemicals are responsible for bioassay responses; and

8. The overlap between the chemical, the exposure and the effects in the laboratory and the effects in the field.

The core objective, however, is to **test the 13 hypotheses, developed by the AETE Committee, at as many of the four selected mine sites as possible (Table 1.1)** The hypotheses are more specific questions about the ability or relative ability of different monitoring tools to answer the four general questions (above) about mine effects.

These 13 hypotheses can be categorized as follows:

- ***Sediment Monitoring***: evaluation of sediment toxicity testing tools (test types) as to their relative ability to detect linkages between mine exposure and sediment toxicity (H1);

- ***Biological Monitoring (in Fish)***: evaluation of tissue biomonitoring tools (measurement types) as to their ability to detect linkages between mine exposure and tissue contamination (H2 to H4); and evaluation of population/community biomonitoring tools (measurement types) as to their ability to detect linkages between mine exposure and ecological response (H5 to H8); and

- ***Integration of Tools***: evaluation of various monitoring tools as to their relative ability to detect relationships between specific measures of mine exposure and specific biological response measures, or between sediment toxicity and benthic community response measures (H9 to H12); and evaluation of effluent toxicity testing tools (test types) as to their ability to detect relationships between effluent toxicity and population/community response measures (H13).

Twelve of the 13 hypotheses (all except H13) were tested at the Mattabi Mine site. In addition, it was desired to evaluate an overall "sediment quality triad" hypothesis, that addresses whether mine-related contaminants appear to be causing biological responses.

TABLE 1.1: HYPOTHESES TESTED IN 1997. AETE FIELD PROGRAM
(Hypotheses in bold print were tested at Mattabi)

Sediment Monitoring **H1. Sediment Toxicity:** **H:** ***The strength of the relationship between sediment toxicity responses and any exposure indicator is not influenced by the use of different sediment toxicity tests or combinations of toxicity tests.***
Biological Monitoring - Fish **H2. Metals in Fish Tissues (bioavailability of metals):** **H:** ***There is no difference in metal concentrations observed in fish liver, kidney, gills, muscle or viscera.***
H3. Metallothionein in Fish Tissues: **H:** ***There is no difference in metallothionein concentration observed in liver, kidney, gills, viscera***
H4. Metal vs. Metallothionein in Fish Tissues: **H:** ***The choice of metallothionein concentration vs. metal concentrations in fish tissues does not influence the ability to detect environmental exposure of fish to metals.***
H5. Fish - CPUE: **H:** ***There is no environmental effect in observed CPUE (catch per unit effort) of fish.***
H6. Fish (or Benthic) - Community: **H:** ***There is no environmental effect in observed fish (or benthic) community structure.***
H7: Fish - Growth: **H:** ***There is no environmental effect in observed fish growth.***
H8. Fish - Organ/Fish Size: **H:** ***There is no environmental effect in observed organ size (or fish size, etc.).***
Integration of Tools **H9.* Relationship between Water Quality and Biological Components:** **H:** ***The strength of the relationship between biological variables and metal chemistry in water is not influenced by the choice of total vs. dissolved analysis of metals concentration.***
H10. Relationship Between Sediment Chemistry and Biological Responses: **H:** ***The strength of the relationship between biological variables and sediment characteristics is not influenced by the analysis of total metals in sediments vs. either metals associated with iron and manganese oxyhydroxides or with acid volatile sulphides.***
H11. Relationship Between Sediment Toxicity and Benthic Invertebrates: **H:** ***The strength of the relationship between sediment toxicity responses and in situ benthic macroinvertebrate community characteristics is not influenced by the use of different sediment toxicity tests, or combinations of toxicity tests.***
H12.* Metals or Metallothionein vs. Chemistry(receiving water): **H:** ***The strength of the relationship between the concentration of metals in the environment (water) and metal concentration in fish tissues is not different from the relationship between metal concentration in the environment and metallothionein concentration in fish tissues.***
H13. Chronic Toxicity - Linkage with Fish and Benthos Monitoring Results: H: *The suite of sublethal toxicity tests cannot predict environmental effects to resident fish performance indicators or benthic macroinvertebrate community structure.*

* Results of H9 and H12 to be interpreted with caution at Mattabi, owing to study design limitations.

1.2 Site Description

Mattabi Mines Ltd. (Mattabi) operated a copper-lead-zinc open pit and underground mining operation and concentrator complex between 1972 and 1991, approximately 80 km northeast of Ignace. As illustrated in Figure 1.1, the mine site is located in northwestern Ontario, between Sturgeon Lake and Bell Lake. The mine was developed between 1970 and 1972 and produced ore between 1972 and 1988. Milling operations continued until 1991. Operations also included mining at the F-Group open pit (development 1979-1980, production 1980-1984) 9 km west of the Mattabi Mine site and the Lyon Lake Mine (Lyon Lake Division; development 1974-1980 production 1980-1991), about 13 km east of the site. Ore from Lyon Lake and F-Group was milled at the Mattabi concentrator.

Rehabilitation activities continue at Mattabi, and effluent continues to be generated from runoff and acid rock drainage (ARD), and is treated with lime addition, polished and discharged to the environment (Bell Creek).

Various studies have been undertaken to document environmental conditions around the Mattabi Mine site, principally by the Noranda Technology Centre (NTC) and Beak Consultants Limited (NTC and BEAK, 1993).

The most recent study by the NTC, completed in 1989, provides an extensive evaluation of physical, chemical and biological conditions in Bell Creek, Bell Creek Bay, Mine Creek, Mine Creek Bay and various inland water bodies potentially affected by Mattabi operations (NTC and BEAK, 1993). The 1989 NTC study also summarizes the results of previous studies carried out by NTC between 1971 and 1987. Limited (fish) data were collected by NTC in 1989 and were reported in the NTC and BEAK study (1993). Conclusions of the 1989 study included the following (among several others):

- The impact of the tailings and polishing pond area on Bell Creek water, sediments and benthos was significant, primarily within the reach extending 6 km downstream of the polishing pond discharge, and some recovery was evident upstream of the mouth of Bell Creek into Bell Creek Bay (Sturgeon Lake). The principal contaminants were zinc (Zn), copper (Cu), lead (Pb) and cadmium (Cd).

- The impact of the Mattabi Mine on Mine Creek Bay (Sturgeon Lake) sediment metal concentrations (Zn, Cu, Pb and Cd) was limited to an approximate 400 m

radius from the outlet of Mine Creek. The benthic community within this area showed some impairment from contamination.

- Sediments in No Name Lake were more contaminated with heavy metals than any of the other lakes and streams affected by the Mattabi Mine, owing to the effects of runoff and seepages affected by ARD.

NTC and BEAK (1993) undertook supplementary environmental work in 1991 and 1992 to fill some information gaps remaining after the 1989 study, and to collect additional information for the purposes of modelling the environmental improvements resulting from reclamation of the Mattabi Mine site.

Water quality conditions showed that concentrations of zinc (Zn) and iron (Fe) were elevated in Bell Creek downstream of the Mattabi site. Metal concentrations in sediments of Bell Creek, No Name Lake and Mine Creek Bay in 1991-92 were consistent with those reported in 1989, with contamination extending to a depth of about 10 cm below the sediment surface. The most contaminated sediments were found in No Name Lake. Of the five sediment samples tested by BEAK for toxicity, none tested positively for toxicity to amphipods (*Hyalella*), but the most contaminated sediments from No Name Lake and Mine Creek Bay caused lethal and sublethal effects in midge (*Chironomus*) larvae.

The 1991-92 benthic community characteristics were similar to those reported by the NTC in 1989. Benthic densities and diversities were reduced in Bell Creek downstream of Mattabi relative to the upstream reference sites. The greatest degree of impairment was observed in Mine Creek Bay and No Name Lake.

Fish sampling demonstrated that suckers and minnows successfully spawn in Bell Creek (NTC and BEAK, 1993). Fish catch records provided by NTC indicated that white sucker were relatively abundant in Bell Creek downstream of the Mattabi site in the 1989 study.

Discussions with Mr. Al Scott of Mattabi Mines prior to the AETE field survey indicated that total zinc concentrations measured downstream of the mine site in Bell Creek remained about 0.15 to 0.2 mg/L in the fall of 1997, and continues to remain elevated and is representative of average levels reported in 1990 and 1991.

Mattabi discharges treated effluent into Bell Creek, a slow-flowing, low gradient stream flowing northward from Bell Lake to Sturgeon lake. Rapids occur in the creek downstream of Bell Lake and upstream of Sturgeon Lake, and may act to discourage regular movement of fish from the reach adjacent to Mattabi to either of the lakes. The creek widens to 1 km downstream of the mine and conditions are more lacustrine than riverine, with mean depths of up to 4 m (but typically $\leq$2 m). Bell Creek sediments are soft and metal-enriched downstream of Mattabi.

2.0 STUDY DESIGN

2.1 General Background

No preliminary study design was developed by EVS *et al.* (1997) for the Mattabi Mine site, as this mine was not originally considered for the 1997 AETE field program. This is in part because the original site selection for the field program considered only operating mines. However, the continued production of treated effluent, combined with the occurrence of nearby sediments enriched in metals and populations of large fish in metal-enriched Bell Creek, afforded good conditions for testing of several AETE hypotheses.

All 13 AETE hypotheses appeared testable at the Mattabi Mine site. However, in light of the fact that H13 (effluent toxicity) was tested at the three other 1997 field sites, effluent toxicity was not studied at Mattabi. This decision was made to allow for sufficient project budget to test the 12 remaining hypotheses.

2.2 General Considerations

In general, sampling is carried out in relation to a point source discharge in order to permit testing of hypotheses about the environmental effect of the discharge. Sampling is carried out both above and below the source (Control versus Exposed). To the extent possible, it is desirable to space the "below discharge" samples at exponentially increasing distances, because most dilution/mixing models assume exponential decay models. That is, a contaminant will decrease in concentration by a given amount over each order of magnitude increase in distance from the discharge (see Figure 2.1). When monitoring mine discharges, the nature of the receiving stream will often cause this ideal to be impossible to achieve, especially where tributary streams produce a stepwise dilution of effluent, or when dilution occurs rapidly (e.g., a stream discharging into a large lake). This latter condition prevails in Bell Creek.

There are many possible field study designs for monitoring of mining discharges and testing of the hypotheses, which can be put into three basic categories (Figure 2.2, Types A, B, C). The difference between the first two (Type A versus Type B or C) is driven by site differences (e.g., stepwise (Type A) versus more continuous dilution patterns (Types B and C)), whereas the difference between the Type B and Type C is driven by the biota being

sampled. For example, because of the sessile nature of some benthos and the limited mobility of some forage fish, allow for replicate sampling in a small area (Type B) with the primary design constraints being hydrology and habitat. For larger more mobile fish, sampling would be carried out over a larger area to ensure the groups of fish are not mixing and are distinct from one another, possibly necessitating a Type C design. Alternatively, a Type A design might be used for large fish, using individual fish rather than stations as replicates.

The ideal situation for testing hypotheses for the 1997 field evaluation is a Type B study design which is a combination of easy-to-sample biota and a site which can be sampled with a gradient design approximating that described above. This provides for:

- a gradient design permitting regression/correlation analysis of the impact pattern along the stream below the discharge, and of possible cause-effect relationships between chemical and biological variables; and
- replication at locations so that testing in an Analysis of Variance (ANOVA) design is possible.

Due to the natural site characteristics at Mattabi, the Type B study design could be implemented for benthos and sediment sampling, but not for large fish sampling.

The other two types of study design (Types A and C) sacrifice either one or the other of the above two attributes (i.e., a gradient design with replication at each location). For Type A, the nature of the site precludes a gradient design (e.g., Bell Creek for water-related exposures). Replicate samples are taken at an "above"="Control" location, and at a "near-field"="High Impact" and at a "far-field"="Low Impact" location. This does not allow one to model the pattern of impact below the discharge, but an ANOVA for testing impact-related hypotheses is easily done. In some cases, two levels of exposure (near- and far-field) cannot both be reasonably sampled due to site conditions, and the Type A configuration is simplified to a reference/exposure (R/E) or control/impact (C/I) design. The C/I design is necessary for testing of fish-related hypotheses in Bell Creek, owing to a general absence of a downstream water quality gradient in Bell Creek (no significant downstream dilution sources entering the creek) and to the mobility of large fish.

For a Type C study design (i.e., gradient design with no replication), one can model the pattern of impact below the discharge but the only possible hypothesis testing is that associated with simple regression analysis. However, there still needs to be a gradient in contaminant levels for this type of design. This type of study design was not used at any of the mine sites used for the 1997 field evaluation program.

Finally, it is necessary to select an appropriate sampling effort and (apart from the above "basic types of design" considerations) to allocate the effort appropriately to above versus below discharge areas, to locations within areas, and to replicates within locations. For the AETE program, it was determined by the AETE Committee that a total sampling effort per mine site of 20 to 25 field samples provided a reasonable balance between feasibility and cost and statistical power and robustness (EVS *et al.*, 1997). The following is based on that total effort allocated to Mattabi.

2.3 Design at Mattabi

Sampling Areas

Much of the study design at Mattabi is of the second type in Figure 2.2 (Type B). This is based on the existence of a sediment chemistry gradient between No Name Lake and central Mine Creek Bay, all within a generally similar benthic habitat type (shallow depth, soft, organic-rich sediments). This gradient is believed to be attributed to the effects of ARD in the drainages of these watersheds. The study design for Mattabi allowed for the collection of sediments for chemical and toxicity testing, as well as for benthic community characterization, at three stations within each of seven areas (Figures 2.3a and 2.3b). This includes five areas along the sediment chemistry gradient and two reference areas - one in Tag Lake (Figure 2.3b), a small unimpacted waterbody similar in other characteristics to No Name Lake, and Peterson Cove, a Sturgeon Lake embayment remote from Mattabi but otherwise similar to Mine Creek Bay (Figure 2.3a). All stations were located at water depths of about 1.0 to 2.0 m.

The exposure gradient in Bell Creek is not clearly defined and waterborne metal concentrations change little with distance downstream of the mine. Exposure and reference sites for collection of fish included a downstream exposure area (Figure 2.4a), whereas a reference area was chosen on the English River (Figure 2.4b). A reference area separate from Bell Creek was chosen because large fish upstream of Mattabi in Bell Creek (e.g., northern pike, white sucker) are able to freely

move downstream and their exposure history would be uncertain. This reference area was selected after consultation with local Ministry of Natural Resources (MNR) biologists and examination of candidate areas at the beginning of the field program.

Water samples were collected at each fish sampling station and each benthic sampling area for chemical characterization.

2.4 Statistical Power

The statistical power of the study design was evaluated using the Borenstein and Cohen (1988) computer code for power analysis. In Bell Creek and the English River (H2, H3 and H4), the total sampling effort of 30 fish distributed equally among two groups (reference and exposure areas) is sufficient to expect that an effect size (average difference between groups) of two within-group standard deviations could be detected with a power of 0.8 or better (i.e., chance of false-negative conclusion (beta) less than 0.2) using a significance criterion based on a chance of false-positive conclusion (alpha) less than 0.05. The total sampling effort of 40 fish (H7, H8, H9) distributed equally among two groups (reference and exposure areas) is sufficient to expect that an effect size (average difference between groups) of one within-group standard deviation could be detected with a power of 0.8 or better using a significance criterion based on an alpha of less than 0.05. The total sampling effort of 8 sampling stations for fish (H5) distributed equally among two groups (reference and exposure areas) is sufficient to expect that an effect size (average difference between groups) of three within-group standard deviations could be detected with a power of 0.8 or better using a significance criterion based on an alpha of less than 0.05.

In the lake habitat, the total sampling effort of 21 sampling stations for benthos, sediment chemistry and toxicity (H1 and H6) equally distributed among seven groups (two reference, 5 exposure) is sufficient to expect that an effect size of two within-group standard deviations could be detected with a power of 0.8 or better using an alpha less than 0.05.

The absolute difference indicated by the one to three standard deviations will vary from one monitoring parameter (effect measure) to another.

For H9 and H12, with a total of eight stations (reference and exposure), it should be possible to detect strong chemistry-biology-toxicity correlations (those that exceed r=0.7;

power=0.8). For H10 and H11, with a total of 15 stations (using only exposure stations), it should be possible to detect strong chemistry-biology-toxicity correlations (those that exceed r=0.7; power=0.8).

3.0 FIELD AND LABORATORY METHODS

3.1 Sampling Time and Crew

The field survey was conducted at Mattabi from 15 to 23 October 1997. The field crew consisted of Robert Eakins and Alan Burt of Beak International Inc., and Don Sinclair and Ryan Benson of Golder Associates Ltd.

3.2 Sampling Station Characterization

Sampling stations for the Mattabi program are illustrated in Figures 2.3a and 2.3b, and Figures 2.4a and 2.4b. Habitat information and station coordinates are provided in Appendix 2.

Sampling areas included five exposure sediment/benthic sampling areas (MME1 to MME5) downgradient of Mattabi in No Name Lake and Mine Creek Bay. Reference areas for these exposure areas consisted of MMR1 in Peterson Cove (Sturgeon Lake), and MMR2 in Tag Lake south of Mattabi. These reference areas were representative of habitat conditions in Mine Creek Bay and No Name Lake, respectively. Three stations were sited within 20 m of a marker buoy located in each area, providing three replicates within each area. Global Positioning System (GPS) coordinates were recorded for each station.

Four fish exposure area stations were sampled in the middle reach of Bell Creek. Each was located in the zone of historical influence from the mine, including elevated levels of metals in water and sediment. Overall, these stations are all exposed to mining effects to similar degrees, and are sufficiently close to one another such that large fish may be expected to move freely among all exposure stations. Although exposed fish may move upstream and vice versa, most of the habitat freely used by fish is affected, with partial habitat barriers (rapids) further upstream and downstream discouraging routine movement of large fish further upstream and downstream (i.e., to Bell Lake on Sturgeon Lake).

The four fisheries reference stations were located in similar proximity to one another in the English River in an area of habitat similar to Bell Creek. Habitat in both fish sampling locations was characterized as wide, slow-moving, soft-bottomed streams, with depths of typically ≤ 2 m.

3.3 Sampling Effort

The numbers and distribution of each type of sample collected at Mattabi are summarized in Table 3.1. Variable numbers of fish collected at each station reflect the presence/ absence and abundances of white sucker and northern pike.

3.4 Water Chemistry

Detailed field sampling procedures, including water quality sampling procedures, are outlined in Annex 1 (provided as a separate document) and summarized in this section.

3.4.1 Field

Most of the 15 water quality samples (with one exception) were collected on 23 October 1997. Samples from the reference site in Peterson Cove were collected on 22 October 1997. An additional sample was collected from the treated effluent discharge channel. Sampling was conducted under dry weather conditions and without significant rainfall during the previous three days. Samples were collected for laboratory analysis of:

- total and dissolved metals (Al, Sb, As, Ba, Be, Bi, B, Cd, Ca, Cr, Co, Cu, Fe, Pb, Mg, Mn, Hg, Mo, Ni, K, Se, Ag, Sr, Ta, Sn, U, V, B and Zn); Zn, Cu, Pb, and Cd are most relevant at Mattabi;
- nutrients (nitrate, nitrite, ammonia, P);
- major ions (including sulphate and ion balance);
- acidity, alkalinity, hardness, specific conductance;
- pH;
- colour;
- dissolved organic and inorganic carbon;
- solids (total suspended and dissolved); and
- turbidity.

In addition to samples collected for laboratory analysis, field determinations were made of specific conductance, water quality, pH and dissolved oxygen, with results recorded in field records. All field measurements were made on-site using calibrated meters.

TABLE 3.1: SUMMARY OF SAMPLES OBTAINED AT MATTABI

	Type of Sample					
Sampling Locations	Chronic Toxicity	Sediment Benthos Toxicity	Water	Fish for Tissue Analysis	Fish Community	Fish for Measurement
Each of 2 Reference Lake Areas*	--	3 stations	2	--	--	--
Each of 5 Exposure Lake Areas**	--	3 stations	5	--	--	--
Bell Creek	--	--	4 stations	15 northern pike 15 white sucker	4 stations	sucker - 20 males, 21 females pike - 15 males, 20 females
English River (Reference)	--	--	4 stations	16 northern pike 16 white sucker	4 stations	sucker - 21 males, 21 females pike - 21 males, 29 females
Total No. of Samples	--	21[1]	15[2]	62[3]	8[4]	168[5]

[1] Each benthic sample is a composite of five grabs; water conductivity and pH measured at each station.

[2] Water sampling stations correspond to sediment sampling areas and fish sampling stations.

[3] White sucker and northern pike in Bell Creek and English River (reference). Tissues analyzed include gill, kidney, liver (metallothionein and metals) and muscle (metals only) for each fish.

[4] These collections produce the fish for analysis and measurement as well as CPUE (catch-per-unit-effort) and taxa for the station.

[5] Fish measurements included fork length, weight, liver weight, gonad weight and fecundity.

* Reference areas include Tag Lake and Peterson Cove (Sturgeon Lake).

**Exposure areas include No Name Lake (highest exposure) plus four above-background areas in Mine Creek Bay.

All samples were placed on ice in coolers immediately after collection, and were transferred to a refrigerator prior to field processing. All samples requiring analysis without chemical preservation were kept chilled until delivery to the laboratory.

Sample containers, filtration and sample preservation procedures are identified in Annex 1, and include use of high density polyethylene containers confirmed free of measurable metal contamination, ultrapure nitric acid and de-ionized distilled water also confirmed by the lab to be free of measurable metal contamination (for field, trip and filter blanks), and a filtration procedure using polypropylene-stoppered syringes with 0.45 micron syringe-filters. All sample preparation was carried out in a clean indoor work space.

Quality assurance/quality control procedures followed in the field included collection of sample duplicates and preparation of trip blanks (Appendix 1). Unfortunately, field blanks and filter blanks for Mattabi were prepared but not analyzed owing to miscommunication with the chemistry laboratory.

3.4.2 Laboratory

All water samples were forwarded to the analytical laboratory (Philip Analytical Services Corporation, Burlington and Mississauga, Ontario) within four days of collection. Procedures used for laboratory analysis are summarized in Table 3.2.

Results of QA/QC analyses indicated no apparent contamination of samples with key metals, based on data from field duplicates and the trip blank. Dissolved and total metal results for field samples did not indicate any significant contamination by dissolved metal sample preparation (i.e., dissolved metals were generally □ total metals), although field and filter blanks were unavailable and thus could not be used to support this conclusion. Data quality appeared adequate for the testing of hypotheses at Mattabi.

3.5 Sediment Chemistry

Annex 1 provides more detail on procedures followed in the field for the collection and handling of sediment samples, which are summarized below.

At each station, samples for benthos were collected prior to sampling for sediment chemistry and toxicity to prevent disruption of the benthic community. For both chemistry and benthic sampling,

Table 3.2: LABORATORY METHODS AND BOTTLE/PRESERVATIVE PROCEDURES USED IN WATER SAMPLE ANALYSIS

(as provided by Philip Analytical Services)

Parameters	Method	Bottle Requirement	Preservative Type	Max. Holding Time
Acidity	Standard Methods (17th ed.) No. 2310B U.S. EPA Method No. 305.1	250 ml Bottle Glass	no preservative	14 days
Alkalinity	Standard Methods (17th ed.) No. 2320	250 ml Bottle Glass	no preservative	14 days
RCAP Calculations Total Dissolved Solids(Calculated) Hardness(as $CaCO_3$) Bicarbonate(as $CaCO_3$, calculated) Carbonate(as $CaCO_3$, calculated) Cation Sum Anion Sum Ion Balance	MDS Internal Reference Method			
Colour	U.S. EPA Method No. 110.3(Modified) (Reference-Std Methods(17th)2120CMod)	100 ml Bottle Glass	no preservative	48 hours
Specific Conductance	U.S EPA Method No. 120.1	100 ml Bottle Glass	no preservative	28 days
Manual Conventionals for RCP(pH,Turb,Conduct,Color)	U.S. EPA Method No. 150.1, 120.1, 180.1 and 110.3	250 ml Bottle HDPE	no preservative	
Hardness	U.S. EPA Method No. 130.2	250 ml Bottle Glass	no preservative	6 months
Ion Balance		250 ml Bottle HDPE	HNO_3 to pH < 2	14 days
pH, Hydrogen Ion Activity	U.S. EPA Method No. 150.1	100 ml Bottle Glass	no preservative	
Total dissolved Solids	U.S. EPA Method No. 160.1	1 L Bottle Glass	no preservative	7 days
Total Suspended Solids	U.S. EPA Method No. 160.2	500 ml Bottle Glass	no preservative	7 days
Turbidity, UltraViolet	U.S. EPA Method No. 180.1	100 ml Bottle Glass	no preservative	48 hours
RCAP MS Package, 8 Element ICPAES Scan B, Fe, P, Zn, Ca, Mg, K, Na	U.S. EPA Method No. 200.7	125 ml Bottle HDPE 250 ml Bottle HDPE	HNO_3 to pH < 2 no preservative	
ICP-MS 25 Element Scan, Clean Water Package Al, Sb, As, Ba, Be, Bi, Cd, Cr, Co, Cu, Pb, Mn, Mo, Ni, Se, As, Sr, Th, Sn, Ti, U, V, B, Fe, Zn	U.S. EPA Method No. 200.8(Modification)	250 ml Bottle HDPE 125 ml Bottle HDPE	no preservative HNO_3 to pH < 2	
Alkalinity for RCAP Packages 30, 50 and MS	U.S. EPA Method No. 310.2	250 ml Bottle HDPE	no preservative	14 days
Anions for RCAP 50 and MS(Cl,NO_2,NO_3,o-PO_4 & SO_4)	U.S. EPA Method No. 300.0 or U.S. EPA Method No. 350.1, 354.1, 353.1, 365.1 and 375.4.	250 ml Bottle HDPE	no preservative	48 hours
Dissolved Organic Carbon, as Carbon for RCAP	MOE Method No. ROM - 102ACE(Modified)	100 ml Bottle Glass	no preservative	3 days
Ammonia for RCAP Packages 30, 50 and MS	ASTM Method No. D1426-79 C	100 ml Bottle Glass	H_2SO_4 to pH < 2	28 days

Table 3.2: LABORATORY METHODS AND BOTTLE/PRESERVATIVE PROCEDURES USED IN WATER SAMPLE ANALYSIS

(as provided by Philip Analytical Services)

Parameters	Method	Bottle Requirement	Preservative Type	Max. Holding Time
	Refer - Method No. 1100106 Issue 122289	250 ml Bottle HDPE	no preservative	
Organic Nitrogen(TKN - NH_3)	U.S. EPA Method No. 350.1 U.S. EPA Method No. 351.1	250 ml Bottle Glass	H_2SO_4 to pH < 2	28 days
Mercury, Cold Vapour AA	U.S. EPA SW846 Method No. 7470A Standard Methods(18th ed.) No. 3112B	100 ml Bottle Glass	HNO_3 to pH < 2 + 5% K_2Cr_2O7	7 days

the boat (or sampling personnel) were re-positioned between sediment grabs to minimize any potential for sampling of disturbed sediments. Finally, sediment areas were sampled in a sequence from areas identified to be less impacted to more impacted to minimize the potential of cross-contamination between sites.

Sediment samples were collected from three stations in each of seven areas along the sediment chemistry gradient. Samples were collected manually using a standard stainless steel petite Ponar grab. Sediments were collected from depths ranging from 1 to 2 m. Ten to fifteen grab samples were collected at each station depending on the quantity of material retrieved in each grab. Sediment pH and redox potential were measured from several minimally disturbed sediment grabs at each station as the composite samples were collected.

Upon retrieval of the grab, surface water was allowed to run off before the sediment was placed into a plastic tub. The top 2 to 3 cm of sediment was collected using a stainless steel spoon and placed into a 20-L bucket with a polyethylene liner. This procedure was repeated with each grab and new material was thoroughly mixed with the previous material until at least eight litres of sediment per station had been collected. Subsamples of the homogenized sediment sample were dispensed into appropriate sample containers.

Three different types of sediment samples were collected for analysis from each site:

- a sample for "total" metals analysis, based on a nitric acid/hydrogen peroxide extraction procedure;
- a sample for "partial" metals analysis using a hydroxylamine hydrochloride procedure which is designed to solubilize amorphous Fe and Mn oxyhydroxides, along with their associated trace metals; and
- a sample for analysis of Acid Volatile Sulphide (AVS) and Simultaneously-Extracted Metals (SEM).

In addition, two field duplicate samples were collected from selected locations for total metals determination using extraction with aqua regia, to confirm the comparability of results using aqua regia and nitric acid/hydrogen peroxide extractions. Subsamples for partial metal extraction were collected by filling half a 500-mL sample bottle with sediment, which was then topped with a layer of site water. These samples were frozen at the end of each sampling day. Subsamples for SEM/AVS analyses were placed into a 250 mL whirl-pak bag, and then into a 1-L jar once the air had been removed from the bag. The 1-L jar was then

filled with sediment so that the whirl-pak bag was surrounded by sediment which prevented exposure to air.

Samples for chemical analysis were forwarded to Philip Analytical Services Corporation. Analyses included metals (total and partial), moisture, bulk density, Munsell colour, total organic carbon (TOC), loss-on-ignition (LOI), grain size and SEM and AVS. Munsell colour, bulk density and percent moisture were done by BEAK staff.

Quality assurance/quality control procedures in addition to routine lab QA/QC included collection of hidden duplicate samples for metal analysis.

3.6 Sediment Toxicity

Sediment samples for toxicity testing were collected from the five exposure areas and two reference areas. A minimum of seven litres of sediment was collected from each of the three stations located within each of the seven areas described in the previous subsection, and were placed in 20-L plastic food-grade buckets with polyethylene bag liners.

Toxicity tests conducted on each sample included: *Hyalella azteca* survival and growth (Environment Canada, 1996 Draft Method); *Chironomus riparius* survival and growth (Environment Canada, 1997 Draft Method); and *Tubifex tubifex* survival and reproduction (ASTM E1384-94A, 1995). *Chironomus* and *Hyalella* tests were conducted at BEAK's toxicity testing laboratory in Dorval, Quebec, whereas the *Tubifex* tests were completed at the National Water Research Institute, Environment Canada, in Burlington, Ontario.

3.7 Benthic Invertebrates

3.7.1 Field

Benthic invertebrates samples were collected from each of three stations in each of seven sediment sampling areas. At each station, five petite Ponar grab samples were collected from water depths of 1 to 2 m and pooled to produce one sample. Each of the five grab samples was sieved using a 250 µm mesh screen prior to preservation to a minimum level of 10% buffered formalin. All samples were collected by the same two field crew members.

3.7.2 Laboratory

All samples were processed jointly by the BEAK Benthic Ecology Laboratory and by Zaranko Environmental Assessment Services (ZEAS), Guelph, Ontario. Both laboratories followed the same laboratory protocols summarized below.

In the laboratory, samples were inspected to insure that they were adequately preserved and correctly labelled. Samples were then stained to improve the sorting recovery.

Prior to detailed sorting, the samples were washed free of formalin in a 250 μm sieve under ventilated conditions. The benthic fauna and associated debris were then elutriated free of any sand and gravel. The remaining sand and gravel fraction was closely inspected for any of the denser organisms, such as Pelecypoda, Gastropoda and Trichoptera with stone cases that may not have all been washed from this fraction. The remaining debris and benthic fauna after elutriation were washed through 500 μm and 250 μm sieves to standardize the size of the debris being sorted and facilitate a minimum of 95% recovery of benthic fauna.

All benthic samples were processed with the aid of stereomicroscopes. A magnification of at least 10X was used for macrobenthos (invertebrates >500 μm) and 20X for meioinvertebrates (invertebrate size >250 to <500 μm). Benthos was sorted from the debris, enumerated into the major taxonomic groups, usually order and family levels and placed in vials for more detailed taxonomic analysis.

Benthic invertebrates were most commonly identified to the lowest practical level, genus or species for most groups. The level to which each group was identified and the taxonomic keys that the identification were based on are provided in Appendix 5.

To assess benthic data quality, subsampling error was determined for both density and number of taxa in 10% of the samples that were subsampled. Ten percent of sorted samples were also resorted by an independent taxonomist to ensure 95% recovery of all invertebrates.

A voucher collection or reference collection of benthic invertebrate specimens was compiled. This is a collection of representative specimens for each taxon so that there can be continuity in taxonomic identifications if different taxonomists process future samples. The voucher collection will be maintained at BEAK. The BEAK and ZEAS Benthic Ecology Laboratories also maintain master reference collections of all taxa which have been identified by the labs.

The specimens selected for the voucher collection were preserved such that they will remain intact for many years. Chironomids and oligochaetes remain on the initial slides and representatives of each taxon were circled with a permanent marker and labelled. All other species were preserved in 80% ethanol in separately labelled vials. Each vial contains a 3% solution of glycerol to prevent spoilage of the fauna if the vials accidentally dry out.

3.7.3 Chironomid Deformities

In the last decade there has been considerable attention paid towards the use of chironomid mouth part deformities to monitor contaminant effects. Previous studies have shown that the incidence of chironomid deformities (especially in *Chironomus*) can be associated with contaminated sediments.

For the 1997 study, all mounted chironomid specimens from each site were scored for mandible and mentum abnormalities. These data were not used in the testing of specific hypotheses, but are discussed briefly in Section 4.0.

3.8 Fish

3.8.1 Collection and Sample Processing

Fish were collected at each of the four Bell Creek and four English River (reference) stations using experimental monofilament gillnets, consisting of 150-foot panels with mesh sizes 1.5, 2, 2.5, 3, 4 and 5 inch stretch mesh. A minimum of two overnight sets was completed at each station.

All fish captured were identified and enumerated, and measurements were taken of total length, fork length and body weight. Live specimens of species other than northern pike and white sucker were released after these measurements were taken. Once the target numbers of northern pike and white sucker were obtained (20 males and 20 females), only non-viable specimens of these species were retained. Viable excess catch of sentinel species were released.

Records of catch-per-unit-effort (numbers - CPUE, biomass - BPUE) were maintained for each gillnet set by accounting for numbers and biomass of fish, as well as the duration of gillnet sets.

Viable northern pike and white sucker for tissue metallothionein and metal analysis were maintained alive (requirement for MT analysis) in plastic garbage pails containing site water and were transported to shore for processing. Fish processed for tissue analyses were first processed for biological measurements including fork length, total length, body weight, age, liver weight, gonad weight and fecundity. Dead northern pike and white sucker specimens were processed for biological measurements only.

Samples for tissue analysis were dissected from freshly euthanized northern pike and white sucker specimens. These dissections included removal of the entire liver, kidney and gills (including gill arches) from each specimen and placement (after liver weight determination) in labelled, plastic bags in direct contact with dry ice. A small (generally ≤50 g) boneless, skinless dorsal fillet of muscle tissue was also collected from each specimen sampled for tissue analysis. All fish tissues were kept on dry ice throughout the field program and during shipment to the Department of Fisheries and Oceans laboratory (Dr. J. Klaverkamp) in Winnipeg.

Biological measurements on fish were carried out according to procedures outlined in Table 3.3. Detailed protocols for these determinations are available in Annex 1.

3.8.2 Tissue Metallothionein and Metal Analyses

All analyses of Mattabi fish tissues were carried out at the Department of Fisheries and Oceans, Freshwater Institute, under the direction of Dr. J. Klaverkamp. Analyses were completed on tissues from 15 to 16 northern pike and white sucker from each of the two sampling areas. Laboratory procedures are as documented by Dr. J. Klaverkamp (Annex 1).

TABLE 3.3: BIOLOGICAL DETERMINATIONS MADE IN NORTHERN PIKE AND WHITE SUCKER SPECIMENS

Measurement	Procedure
Fork Length, Total Length	Fish measuring boards, to nearest 1 mm.
Fish Weight	Calibrated spring scales, to nearest 5 g.
Liver Weight, Gonad Weight	Calibrated electronic balance, to nearest 0.1 g (based on fresh weight).
Fecundity	Measurement of volume (by water displacement) of predetermined number of preserved eggs (formaldehyde), and measurement of preserved ovary volumes, to nearest mL.
Age	Examination of annuli on cleithra (northern pike) and in sections of first large left pectoral fin rays (white sucker), to nearest year.

4.0 DATA OVERVIEW

This section summarizes the major trends for each of the data components (water, sediment, sediment toxicity, benthos and fish), whereas results of hypotheses testing based on the data are presented in Section 5.2.

4.1 Water Chemistry

Selected water chemistry data for Mattabi are summarized in Table 4.1 (total metals and general chemistry) and Table 4.2 (comparing total versus dissolved metals). Detailed data for all parameters and samples are presented in Appendix 3.

Effluent chemistry was sampled during the water chemistry sampling program at Mattabi, with results presented in Appendix 3 ("discharge" sample). Results show the effluent to be enriched with zinc, calcium and sulphate in particular. Mean effluent quality conditions for October 1997 at Mattabi, as provided by Mr. Al Scott (Mattabi Mines) (pers. comm.), are as follows:

- pH: 8.79
- TSS: 3 mg/L
- As: 0.042 mg/L
- Cu: 0.010 mg/L
- Fe: 0.304 mg/L
- Pb: 0.025 mg/L
- Ni: 0.020 mg/L
- Zn: 0.32 mg/L

QA/QC data associated with water chemistry analyses are provided in Appendix 1. Results of the QA/QC program indicate that the quality of Mattabi water chemistry data are adequate for testing of hypotheses, with data quality objectives for replicate samples met for key metals.

Concentrations of zinc, copper, cadmium and lead in water were elevated in Bell Creek (exposure) samples downstream of Mattabi relative to conditions in the English River (reference). Of these, only zinc occurred at concentrations greater than *Canadian Water Quality Guidelines* (CWQGs), with concentrations of 0.031 to 0.061 mg/L for total zinc. The sample from Station MME6 in Bell Creek showed lower metal concentrations than those from further downstream, apparently owing to less complete mixing of Mattabi effluent in the creek at the more upstream site relative to

Table 4.2: Concentration of Total Metal Concentrations versus Dissolved Metal Concentrations Detected in Water Samples Collected at Mattabi Mine Site, October 1997.

			MINE DISCHARGE		REFERENCE STATIONS (LAKE)				EXPOSURE STATIONS (LAKE)								
Parameters	Units	LOQ[1]	Total	Dissolved	MMR1 Total	MMR1 Dissolved	MMR2 Total	MMR2 Dissolved	MME1 Total	MME1 Dissolved	MME2 Total	MME2 Dissolved	MME3 Total	MME3 Dissolved	MME4 Total	MME4 Dissolved	MME5 Total
Aluminum	mg/L	0.005	0.098	0.052	0.005	nd	0.028	0.014	0.102	0.033	0.006	nd	0.008	nd	0.006	nd	0.005
Cadmium	mg/L	0.00005	0.00151	0.00119	nd[2]	nd	nd	nd	0.00185	0.00183	nd	nd	0.00005	nd	nd	nd	nd
Calcium	mg/L	0.1	831	868	13.7	15.3	7.9	8.2	71.5	73.3	13.9	14.8	13.9	14.2	14	14.3	13.9
Cobalt	mg/L	0.0002	0.0021	0.0018	nd	nd	nd	nd	0.0031	0.0034	nd	nd	nd	nd	nd	nd	nd
Copper	mg/L	0.0003	0.0154	0.0128	0.002	0.0008	nd	0.0008	0.0199	0.0133	0.0016	0.0019	0.0019	0.002	0.0016	0.0021	0.0015
Iron	mg/L	0.02	0.21	0.1	0.05	nd	0.13	0.03	0.23	0.08	0.05	nd	0.05	nd	0.06	nd	0.05
Lead	mg/L	0.0001	0.0002	0.0002	nd	0.0002	0.0003	nd	0.0058	0.0021	0.0001	nd	0.0001	nd	0.0001	0.0002	nd
Selenium	mg/L	0.002	0.004	0.004	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Zinc	mg/L	0.001	0.206	0.094	0.006	0.008	0.001	0.002	2.61	2.61	0.014	0.013	0.02	0.016	0.018	0.018	0.012

			REFERENCE STATIONS (RIVER)								EXPOSURE STATIONS (RIVER)						
Parameters	Units	LOQ[1]	MMR10 Total	MMR10 Dissolved	MMR11 Total	MMR11 Dissolved	MMR12 Total	MMR12 Dissolved	MMR13 Total	MMR13 Dissolved	MME6 Total	MME6 Dissolved	MME7 Total	MME7 Dissolved	MME8 Total	MME8 Dissolved	MME9 Total
Aluminum	mg/L	0.005	0.043	0.023	0.052	0.026	0.043	0.023	0.044	0.031	0.026	0.017	0.038	0.027	0.036	0.025	0.036
Cadmium	mg/L	0.00005	nd[2]	nd	nd	nd	nd	nd	nd	nd	0.00005	nd	0.00006	nd	0.00009	nd	0.00006
Calcium	mg/L	0.1	4.8	5	5	4.9	4.9	4.9	4.8	4.9	8.2	8.7	22.3	22.1	25.8	24.8	24.6
Cobalt	mg/L	0.0002	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0003	0.0002	0.0003	0.0003	0.0002
Copper	mg/L	0.0003	0.0005	0.0006	0.0006	0.0009	0.0005	0.0008	0.0005	0.0006	0.0014	0.0017	0.0019	0.0022	0.0019	0.0021	0.002
Iron	mg/L	0.02	0.32	0.2	0.35	0.18	0.3	0.21	0.31	0.2	0.08	0.04	0.1	0.07	0.11	0.07	0.1
Lead	mg/L	0.0001	nd	0.0004	nd	0.0001	nd	0.0001	nd	0.0002	nd	0.0001	0.0009	0.0008	0.0008	0.0007	0.0009
Selenium	mg/L	0.002	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Zinc	mg/L	0.001	nd	nd	nd	0.001	0.001	0.001	nd	0.002	0.031	0.029	0.057	0.055	0.059	0.06	0.061

[1] LOQ - Limit of Quantitation = lowest level of the parameter that can be quantified with confidence

[2] nd - Parameter not detected

Table 4.2: Concentration of Td.

Parameters	Units	LOQ[1]	MME5 Dissolved
Aluminum	mg/L	0.005	0.005
Cadmium	mg/L	0.00005	nd
Calcium	mg/L	0.1	14.2
Cobalt	mg/L	0.0002	nd
Copper	mg/L	0.0003	0.0017
Iron	mg/L	0.02	nd
Lead	mg/L	0.0001	nd
Selenium	mg/L	0.002	nd
Zinc	mg/L	0.001	0.011

Parameters	Units	LOQ[1]	MME9 Dissolved
Aluminum	mg/L	0.005	0.025
Cadmium	mg/L	0.00005	nd
Calcium	mg/L	0.1	25.6
Cobalt	mg/L	0.0002	0.0002
Copper	mg/L	0.0003	0.0022
Iron	mg/L	0.02	0.07
Lead	mg/L	0.0001	0.0007
Selenium	mg/L	0.002	nd
Zinc	mg/L	0.001	0.059

[1] LOQ - Limit of Quantitation = lowest

[2] nd - Parameter not detected

downstream (i.e., MME6 was located immediately offshore of the effluent discharge point). Water hardness, calcium concentration and conductivity were also greater in Bell Creek than at the reference site (or in Sturgeon Lake), reflecting the effects of lime addition at the Mattabi treatment plant.

In addition to the metals noted above, selenium and cobalt concentrations were detectable in treated Mattabi effluent but were lower in concentration in Bell Creek (Se was not detected in river samples; Appendix 3). This indicates that the mine may also be a source of these two metals. Hypotheses relating to water quality effects were tested based on conditions measured in Bell Creek and the English River.

Metal concentrations (total and dissolved Al, Zn, Cu, Pb, Cd) were elevated in No Name Lake (MME1) relative to exposure areas in Mine Creek Bay (Sturgeon Lake; MME2 to MME5), where metal levels were much less elevated relative to reference sites. Of these metals, all except Pb occurred in excess of their respective Canadian Water Quality Guidelines for the protection of aquatic life (Table 4.1). No Name Lake water quality is affected by ARD, whereas ARD sources to Mine Creek Bay have been largely eliminated through site rehabilitation activities. Also, any residual ARD sources to Mine Creek Bay are dispersed owing to the large assimilative capacity of Sturgeon Lake. Accordingly, metal concentrations in Mine Creek Bay were found to meet the Canadian guidelines. These substantially greater metal concentrations in No Name Lake relative to Mine Creek Bay indicate that sediment-related hypotheses, tested using the sediment chemistry gradient from MME1 (No Name Lake) to MME5 (Mine Creek Bay), may be confounded by effects from elevated aqueous metal concentrations at MME1. In particular, zinc concentrations of 2.61 mg/L are close to levels known to be acutely toxic to invertebrates (e.g., LC50 value for *Daphnia magna* about 2.8 mg/L in hard water; U.S. EPA, 1987). The high metal concentrations at MME1 are accompanied by hard water conditions (hardness = 285 mg/L) which may act to modify metal effects relative to those which may occur at water hardness levels of about 45 mg/L in Mine Creek Bay.

The total and dissolved concentrations of selected metals are provided in Table 4.2 and in Figure 4.1, with a complete data set provided in Appendix 3. Dissolved metal concentrations were similar to those for the corresponding total metals for most key metals. Dissolved lead levels were, however, substantially less than total lead levels in the most metal-enriched sample (e.g., No Name Lake). The lower dissolved lead relative to total lead concentrations at MME1 may reflect lead precipitation or coprecipitation with iron hydroxide, as suggested by the substantial total iron concentration in the water (Table 4.2).

4.2 Sediment Chemistry

Fine-grained sediments are present both at stream and lake sampling stations; however, sediments were collected for hypothesis testing only at lake stations. Historical sediment quality data for Bell Creek indicate high but spatially heterogeneous metal concentrations in Bell Creek sediments, whereas those in Mine Creek Bay show a more systematic spatial gradient, facilitating a more effective study design in the latter area. Nonetheless, any biological effects observed in fish in Bell Creek may be attributed to either or both sediment metals and aqueous metals.

Sediment chemistry data are summarized in Table 4.3 (total metals), Table 4.4 (partial metals and Table 4.5 (acid volatile sulphide/simultaneously extracted metals). All raw data are provided in Appendix 4. The total metal concentrations are compared with Canadian Interim Sediment Quality Assessment Values (CISQAV) (Environment Canada, 1995). The TEL (threshold effect level) refers to the concentration below which an adverse effect is likely to rarely occur, while the PEL (probable effect level) refers to the concentration above which one could frequently expect adverse effects. All sediment QA/QC data are provided in Appendix 1.

Total Metal Concentrations and Physical Sediment Characteristics

All sediments were characterized as organic-rich and sandy, although reference area MMR1 in Sturgeon Lake tended to be somewhat coarser in grain size than other areas. Organic (TOC) concentrations were typically 20 to 30%. Sediment colour was classified as black in the exposure gradient, but somewhat different in colour at the reference sites (Table 4.3).

Concentrations of total zinc, lead, cadmium, copper, arsenic, nickel and mercury in sediments exceeded PEL levels at most exposure site stations, with maximum values as much as two orders of magnitude greater than PELs for zinc (Table 4.3 and Figure 4.2). There was a general decreasing trend in concentration from MME1 to MME5 although, for some metals other than zinc, concentrations were greater at MME5 than at MME4. Total metal concentrations were greater at all exposure areas than at either of the two reference areas.

Generally, similar results were obtained in duplicate samples extracted using the hydrogen peroxide/nitric acid leach and conventional aqua regia (Appendix 1).

Geochemical Fractions

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Table 4.3: Selected Sediment Quality Results at Mattabi Mine Site, October 1997. Metals Results Represent Total Metals Analyses.

Parameter	Units	MDL[1]	CISQAV[2] TEL[3]	CISQAV[2] PEL[4]	REFERENCE STATIONS						EXPOSURE STATIONS													
					MMR1-1	MMR1-2	MMR1-3	MMR2-1	MMR2-2	MMR2-3	MME1-1	MME1-2	MME1-3	MME2-1	MME2-2	MME2-3	MME3-1	MME3-2	MME3-3	MME4-1	MME4-2	MME4-3	MME5-1	MME5-2	MME5-3
TOC (Solid)	%	0.1	na	na	26	27	28	34	36	26	20	19	19	23	24	21	28	26	21	23	23	26	25	26	25
Aluminum	mg/kg	1	na	na	5200	3900	3700	2700	3000	3000	5000	11000	12000	6300	6200	6600	6000	6400	5900	5800	4900	6000	6600	6100	4900
Arsenic	mg/kg	0.5	5.9	17	3.5	3.9	3.4	5.4	7.2	7.1	67	190	160	41	32	43	19	26	24	16	4.5	13	14	20	18
Cadmium	mg/kg	0.05	0.596	3.53	1.8	1.6	1.6	0.94	1.2	1.2	60	100	140	43	33	63	36	56	39	18	5	17	5	14	21
Calcium	mg/kg	20	na	na	8680	9020	8856	9563	10010	8956	4187.5	9315	9075	11262.5	11050	10425	11332.5	10752.5	10465	11570	11967.2	11115	11335	11387.5	-
Cobalt	mg/kg	0.2	na	na	3.1	2.7	2.5	3.1	3.2	3.6	36	62	67	36	29	49	20	31	23	14	6.1	8.7	10	11	11
Copper	mg/kg	0.2	35.7	196.6	51	43	42	7.9	8.5	13	1100	2500	2500	1000	1200	1700	930	1300	1100	780	160	530	620	930	890
Iron	mg/kg	20	na	na	6100	4700	4400	3600	4300	3800	15000	33000	36000	16000	14000	17000	13000	15000	13000	12000	10000	10000	12000	13000	12000
Lead	mg/kg	0.1	35	91.3	25	26	24	29	27	36	760	1300	1700	870	600	980	490	670	490	300	55	280	330	540	560
Mercury	mg/kg	0.04	0.174	0.486	0.08	0.09	0.08	0.1	0.09	0.1	1.3	1.3	1.5	0.78	0.53	0.77	0.45	0.59	0.51	0.33	0.12	0.29	0.28	0.46	0.51
Nickel	mg/kg	0.5	18	35.9	19	18	17	5.1	5.6	5.9	50	81	86	50	48	56	34	45	35	37	26	25	24	24	27
Selenium	mg/kg	1	na	na	2.3	3.2	3.2	1.7	2.4	2.2	7	17	17	23	21	24	8.8	16	12	12	2.9	5	4.1	6.2	5.8
Zinc	mg/kg	1	123.1	314.8	450	350	360	78	90	97	20000	42000	45000	15000	13000	21000	11000	14000	13000	6000	2800	27000	26000	14000	11000
Grain Size Analysis																									
Gravel (>2.0 mm - 4.8 mm)	(%)	0.1	na	na	0.2	4.9	3.5	3.4	0.2	2.6	1.1	1.2	0.2	4.1	1.3	4.2	7	6.6	6.6	2.2	2.8	3.1	3.7	6.2	6
Sand (2.0 mm - 0.050 mm)	(%)	0.1	na	na	94.3	88.4	80.9	89.3	4.5	66.5	55.1	68.1	61	66.3	75.6	73.4	70	60.7	78.9	56.2	72	75.7	73.5	57.8	80.9
V. Fine Sand, Silt, Clay (<0.10 mm)	(%)	0.1	na	na	5	7.8	16	7.7	95	31	44	31	39	31	23	22	23	34	15	42	26	22	23	36	13
Silt (0.002-0.050mm)	(%)	0.1	na	na	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Clay (<0.002mm)	(%)	0.1	na	na	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eh	mV		na	na	-98	-80	-80	-11	-18	-11	-180	-165	-150	-151	-90	-137	-180	-132	-76	-163	-106	-90	-125	-130	-132
Bulk Density	g/mL		na	na	0.043	0.040	0.038	0.025	0.023	0.024	0.068	0.067	0.065	0.059	0.061	0.063	0.044	0.048	0.048	0.056	0.049	0.041	0.051	0.054	0.053
Sediment Moisture	%		na	na	95.9	96.1	96.3	97.5	97.8	97.6	93.3	93.4	93.6	94.14	94.22	94.0	95.6	95.2	95.2	94.4	95.3	96.0	95.0	94.8	94.7

[1] MDL - Method detection limit - lowest level of the parameter that can be detected with confidence
[2] CISQAV - Canadian Interim Sediment Quality Assessment Values (Freshwater) (Environment Canada, 1995)
[3] CISQAV - Threshold Effect Level (TEL)
[4] CISQAV - Probable Effect Level (PEL)
na - Guideline values not available
" Silt and clay size fractions could not be readily distinguished owing to sediment consistency (Philip Analytical Services, Personal Communication)
- Denotes values that exceed the Threshold Effect Level (TEL)
- Denotes values that exceed the Probable Effect Level (PEL)

Table 4.3: Selected Sediment Quality Results at Mattabi Mine Site, October 1997. Metals Results Represent Total Metals Analyses.

Parameter	Units	MDL[1]	CISQAV[2] TEL[3]	CISQAV[2] PEL[4]	REFERENCE STATIONS MMR1-1	MMR1-2	MMR1-3	MMR2-1	MMR2-2	MMR2-3	EXPOSURE STATIONS MME1-1	MME1-2	MME1-3	MME2-1	MME2-2	MME2-3	MME3-1	MME3-2	MME3-3	MME4-1	MME4-2	MME4-3	MME5-1	MME5-2	MME5-3
TOC (Solid)	%	0.1	na[5]	na	26	27	28	34	36	26	20	19	19	23	24	21	28	26	21	23	23	26	25	26	25
Aluminum	mg/kg	1	na	na	5200	3900	3700	2700	3000	3000	5000	11000	12000	6300	6200	6600	6000	6400	5900	5500	4900	6000	6600	6100	4900
Arsenic	mg/kg	0.5	5.9	17	3.5	3.9	3.4	5.4	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	41	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	16	4.5	13	14	[illegible]	[illegible]
Cadmium	mg/kg	0.05	0.596	3.53	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Calcium	mg/kg	20	na	na	8680	9020	8856	9563	10010	8956	4187.5	9315	9075	11262.5	11050	10425	11332.5	10752.5	10465	11470	11967.2	11115	11335	11387.5	-
Cobalt	mg/kg	0.2	na	na	3.1	2.7	2.5	3.1	3.2	3.6	36	62	67	36	29	49	20	31	23	14	6.1	8.7	10	11	11
Copper	mg/kg	0.2	35.7	196.6	[illegible]	[illegible]	[illegible]	7.9	8.5	13	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Iron	mg/kg	20	na	na	6100	4700	4400	3600	4300	3800	15000	33000	36000	16000	14000	17000	13000	15000	13000	12000	10000	10000	12000	13000	12000
Lead	mg/kg	0.1	35	91.3	25	26	24	29	27	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Mercury	mg/kg	0.04	0.174	0.486	0.08	0.09	0.08	0.1	0.09	0.1	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	0.33	0.12	0.29	[illegible]	[illegible]	[illegible]
Nickel	mg/kg	0.5	18	35.9	[illegible]	18	17	5.1	5.6	5.9	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Selenium	mg/kg	1	na	na	2.3	3.2	3.2	1.7	2.4	2.2	7	17	17	23	21	24	8.8	16	12	12	2.9	5	4.1	6.2	5.8
Zinc	mg/kg	1	123.1	314.8	[illegible]	[illegible]	[illegible]	78	90	97	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Grain Size Analysis																									
Gravel (>2.0 mm - 4.8 mm)	(%)	0.1	na	na	0.2	4.9	3.5	3.4	0.2	2.6	1.1	1.2	0.2	4.1	1.3	4.2	7	6.6	6.6	2.2	2.8	3.1	3.7	6.2	6
Sand (2.0 mm - 0.050 mm)	(%)	0.1	na	na	94.3	88.4	80.9	89.3	4.5	66.5	55.1	68.1	61	66.3	75.6	73.4	70	60.7	78.9	56.2	72	75.7	73.5	57.8	80.9
V. Fine Sand, Silt, Clay (<0.10 mm)	(%)	0.1	na	na	5	7.8	16	7.7	95	31	44	31	39	31	23	22	23	34	15	42	26	22	23	36	13
Silt (0.002-0.050mm)	(%)	0.1	na	na	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Clay (<0.002mm)	(%)	0.1	na	na	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eh	mV		na	na	-98	-80	-80	-11	-18	-11	-180	-165	-150	-151	-90	-137	-180	-132	-76	-163	-106	-90	-125	-130	-132
Bulk Density	g/mL		na	na	0.043	0.040	0.038	0.025	0.023	0.024	0.068	0.067	0.065	0.059	0.061	0.063	0.044	0.048	0.048	0.056	0.049	0.041	0.051	0.054	0.053
Sediment Moisture	%		na	na	95.9	96.1	96.3	97.5	97.8	97.6	93.3	93.4	93.6	94.14	94.22	94.0	95.6	95.2	95.2	94.4	95.3	96.0	95.0	94.8	94.7

[1] MDL - Method detection limit - lowest level of the parameter that can be detected with confidence
[2] CISQAV - Canadian Interim Sediment Quality Assessment Values (Freshwater) (Environment Canada, 1995)
[3] CISQAV - Threshold Effect Level (TEL)
[4] CISQAV - Probable Effect Level (PEL)
[5] na - Guideline values not available
[6] Silt and clay size fractions could not be readily distinguished owing to sediment consistency (Philip Analytical Services, Personal Communication)
[illegible] - Denotes values that exceed the Threshold Effect Level (TEL)
[illegible] - Denotes values that exceed the Probable Effect Level (PEL)

Partial metal extractions and ratios of acid volatile sulphide and simultaneously extracted metals may potentially be useful indicators of sediment metal bioavailability or sediment toxicity. Thus, these measurements may be of greater value than total sediment metal concentrations as tools to predict biological impact.

Partial Metals

At Mattabi, partial metals represented small fractions of total metals for zinc, copper, cadmium and lead (Table 4.4; Figure 4.2). In the most contaminated sediments, these fractions were about 10% for zinc and <1% for some of the other metals (Cu, Cd). The partial iron fraction was about 10% of total iron found in exposure sediments, indicating that iron hydroxide-bound forms may account for the partial fractions for most metals (i.e., those having partial metal fractions of ≤10%).

Partial metal concentration gradients were weaker than those for total metals or not evident for key metals from MME1 to MME5. This implies that most of the metals present in Mattabi sediments require a more aggressive leach than hydroxylamine hydrochloride to produce dissolution, and that the largest fractions of these metals are not controlled by iron hydroxide coprecipitation or by more readily dissociated particle-metal forms.

Acid Volatile Sulphide (AVS) and Simultaneously Extracted Metals (SEM)

In general, SEM/AVS ratios <1 may reflect non-toxic sediment conditions because some of the key metals (e.g., Ni, Pb, Cu, Cd, Zn) which are often associated with sediment toxicity will be in sulphide forms which reduces their bioavailability. However, it is possible that sediments with SEM/AVS ratios <1 will still be toxic due to the presence of other metals (e.g., arsenic, mercury) which are not included in the SEM analysis.

SEM/AVS ratios >1 often reflect sediments that may be toxic because there is insufficient sulphide to react with the bioavailable metals to make them less toxic. Again, SEM/AVS ratios >1 do not always accurately predict that sediments will be toxic because other factors, such as organic material or clay, will also bind metals, thereby reducing their toxicity.

The SEM/AVS ratio was developed to predict acute sediment toxicity and not necessarily for predicting chronic effects, including effects on the benthic community. However, it is not unreasonable to expect that, if sediments are acutely toxic, there would be some change in the

benthic community structure that reflects this toxicity. Therefore, there may be a correlation between SEM/AVS ratios >1 and effects observed on benthic communities. This correlation is investigated in this report.

SEM/AVS ratios calculated for Mattabi sediments are presented in Table 4.5 and summarized in Figure 4.3. The data indicate spatially variable results, with ratios averaging above 1 for most sampling areas including one of the reference areas. With the exception of one of the three No Name Lake (MME1) sediment samples, there is no clear difference in mean SEM/AVS ratios between reference and exposure areas. At MME1, however, the ratios were more variable among the three sediment samples, with one value greater than at other sites.

Aqua Regia versus Nitric Acid/Hydrogen Peroxide Extraction Method

Two samples (MMR1-1 and MME1-2) were analyzed for total metals after extraction by *aqua regia* to compare with results obtained by analysis using the nitric acid/hydrogen peroxide leach used for total metals analysis throughout this study (Appendix 1). There was relatively little variation in results between the two methods, with Cu, Cd and Pb results differing by ≤12% between the two extraction procedures.

4.3 Sediment Toxicity

Toxicity tests were carried out on the 15 sediment samples collected along the sediment quality gradient (three at each of MME1 to MME5) and from the six reference sediment samples (three each in the two reference areas). The tests included *Chironomus riparius* survival and growth, *Hyalella azteca* survival and growth, and *Tubifex tubifex* survival and reproduction. The principal endpoints in the *Tubifex* test are sublethal reproductive endpoints, because this species is recognized as insensitive in terms of its mortality response to many toxicants. Laboratory report summaries are presented in Appendix 4, with full reports provided in Annex 1.

Survival of all three species showed little apparent response to the sediment quality gradient (Figure 4.4; Table 4.6). Mortality responses occurred in both *Chironomus* and *Hyalella* in both reference and exposure area sediments. The data suggest a greater response in *Hyalella* in the most contaminated sediments (MME1) than at most other locations. This contrasts with sediment toxicity results obtained in 1991-1992, which showed no effects on *Hyalella* but some lethal and sublethal effects in *Chironomus*.

At the sublethal level, reference-exposure differences were similarly absent or at best weakly apparent. Production of young in *Tubifex* appeared to be slightly reduced in MME1 sediments relative to most sediments, but MMR2 reference sediments produced a similar response, suggesting this was not a mine exposure effect. *Hyalella* growth was lowest at MME3 and *Chironomus* growth lowest at MME4 on average, although growth in these sediments was variable within each of the two areas and mean sizes were not substantially different than growth observed in some of the reference sediments.

Toxicity of sediments did not appear to be related to the SEM/AVS ratio, despite the fact that ratio values fell both above and below 1.0 in the dataset (Figure 4.5). Indeed, the fact that significant levels of *Hyalella* and *Chironomus* mortality occurred at SEM/AVS ratios below 1.0 suggest that the SEM/AVS model was ineffective in this instance.

Sediment toxicity also did not appear related to the sum of molar Cd, Cu, Pb and Zn (partial extractions), expressed as fraction of the molar concentration of iron in the partial extractions (Table 4.4; Figure 4.6). A possible relationship here could be that toxicity occurs when heavy metals are present in excess of the molar concentration of Fe, because the excess portions are not coprecipitated with iron. This possible model is somewhat analogous to the model that states that SEM in excess of AVS concentrations can cause toxicity.

4.4 Benthic Invertebrates

Benthic invertebrate data are provided in Appendix 5. Associated QA/QC data are provided in Appendix 1. Data summaries are provided in Table 4.7 and Figure 4.7.

Benthic community trends are apparent in the sediment chemistry gradient, with reduced numbers of taxa in the most exposed stations, as well as changes in the abundances of possible indicator taxa. Common or abundant taxa included Harpacticoida and chironomids, with certain common taxa conspicuously absent or abundant at the most impacted stations. The sensitive taxa included Hydracarina (mites), *Caenis* (mayfly) and *Pisidium* (mollusc), whereas the chironomid

Table 4.7: Benthic Community Indices, Based on Petite Ponar Sampler Collections, Mattabi Mine, October 1997.

Station	Number of Individuals [1]	Number of Taxa	Hydracarina (%)	Caenis (%)	Chironomus (%)	Procladius (%)	Tanypodinae (%)	Chironominae (%)	Pisidium (%)
MME1-1	5442	12	0.00	0.00	4.01	6.69	11.04	9.03	0.00
MME1-2	3240	9	0.00	0.00	0.00	10.13	15.17	11.24	0.00
MME1-3	3813	11	0.47	0.00	1.44	9.55	12.65	4.77	0.00
MME2-1	6006	21	1.82	4.25	10.91	12.13	12.12	41.21	2.42
MME2-2	6152	22	3.55	2.96	7.11	9.47	11.83	51.48	1.18
MME2-3	4222	21	0.85	9.48	6.04	18.98	24.14	45.69	1.72
MME3-1	4659	21	3.14	5.48	16.41	10.16	12.50	35.16	3.13
MME3-2	5387	23	3.38	8.11	10.81	12.16	14.86	32.43	2.70
MME3-3	4623	19	3.94	3.16	11.81	13.39	15.75	32.28	3.15
MME4-1	4950	22	4.40	34.20	7.35	24.63	25.74	16.54	2.21
MME4-2	5041	26	10.13	19.52	7.23	24.20	26.71	17.69	2.53
MME4-3	4805	24	3.42	9.10	17.06	19.70	21.97	26.89	4.17
MME5-1	8199	27	3.55	3.55	11.99	11.99	15.09	54.16	6.22
MME5-2	5824	26	4.38	4.38	10.63	16.25	20.00	41.25	10.00
MME5-3	8919	28	1.64	2.04	6.53	3.68	11.02	49.80	8.98
MMR1-1	22271	30	2.29	0.82	0.00	2.78	2.78	54.25	1.96
MMR1-2	35090	24	3.63	1.87	0.00	2.80	2.80	34.54	1.04
MMR1-3	33734	24	4.99	1.48	0.00	2.62	2.62	31.02	1.08
MMR2-1	6425	23	6.52	3.69	0.00	22.38	22.38	24.93	2.83
MMR2-2	5244	25	6.25	3.13	0.00	12.49	12.50	30.56	4.86
MMR2-3	6497	27	5.88	1.96	0.00	19.62	19.61	24.93	6.72

[1] Number of individuals per m^2 composite sample from a Petite Ponar.

Psectrocladius was abundant and dominant uniquely at MME1 (No Name Lake) stations. These trends do not appear to be attributed to variations in sediment texture or organic content, as all sediments generally appeared similar in the field (No Name Lake similar to Tag Lake, Mine Creek Bay similar to Peterson Cove). To further illustrate this point, the greatest differences in sediment texture were observed between the two reference areas (coarsest and finest) and yet the benthic trends noted above correspond with reference-exposure area differences.

Data on mouth-part deformities and abnormalities in chironomid larvae are presented in Appendix 5. The data suggest no reference-exposure differences in the incidence of anomalies.

4.5 Fish

4.5.1 Fish Health and Community

Detailed biological measurements on all fish captured in the Mattabi survey are presented in Appendix 6.

Several species of fish were captured in Bell Creek (exposure area) and the English River (reference area) (Table 4.8). The two sentinel species (northern pike and white sucker) were relatively abundant in each area, although pike appeared to be more abundant in the reference area and sucker more abundant in the exposure area. Catch-per-unit-effort in terms of numbers and biomass for all fish species combined were not distinctly different between the reference and exposure areas (Table 4.8; Figure 4.8). The principal difference in the fish community between the two areas is the presence of lake whitefish in the English River but not in Bell Creek; this probably represents an effect of habitat differences between Bell Creek and the English River. That is, the English River is a much larger river in general than Bell Creek, and thereby provides suitable habitat for lake whitefish. This habitat difference was less apparent at the local level, because all sampling areas were in wide (generally 100 to 1,000 m in width) reaches having relatively shallow mean depths. Deeper water suitable for whitefish, however, is present near reference areas, whereas habitat barriers (shallow riffles) probably prevent the seasonal movement of whitefish from Sturgeon Lake or Bell Lake into exposure areas in Bell Creek.

Size at age graphs show little apparent differences in growth of white sucker between reference and exposure areas (Figure 4.9; Table 4.9). However, northern pike showed greater growth in the Bell Creek exposure area than in the English River.

Table 4.8: Catch per Unit Effort in the Reference and Exposure Areas. Mattabi Mine, October 1997.

Area	Station	Date/Time Set	Date/Time Lift	Fishing Time (hrs)	Northern Pike (fish/hr)	White Sucker (fish/hr)	Walleye (fish/hr)	Yellow Perch (fish/hr)	Sauger (fish/hr)	Shorthead Redhorse (fish/hr)	Lake Whitefish (fish/hr)	Lake Herring (fish/hr)	All Fish (fish/hr)
Exposure	MM6	10/15/97 15:50	10/16/97 11:20	19.5	0.359	0.359	0.000	0.103	0.000	0.000	0.000	0.000	0.821
		10/17/97 15:15	10/18/97 11:50	20.6	0.049	0.243	0.049	0.049	0.000	0.049	0.000	0.000	0.437
		10/18/97 12:25	10/19/97 10:50	22.4	0.134	0.357	0.000	0.045	0.000	0.045	0.000	0.000	0.580
		10/18/97 12:15	10/19/97 11:25	23.2	0.086	0.129	0.000	0.086	0.000	0.043	0.000	0.000	0.345
			Mean	**21.4**	**0.157**	**0.272**	**0.012**	**0.071**	**0.000**	**0.034**	**0.000**	**0.000**	**0.546**
	MM7	10/15/97 16:20	10/16/97 12:30	20.2	0.099	0.397	0.099	0.149	0.000	0.000	0.000	0.000	0.744
		10/16/97 13:25	10/17/97 14:45	25.3	0.395	0.395	0.000	0.039	0.000	0.039	0.000	0.000	0.868
			Mean	**22.8**	**0.247**	**0.396**	**0.050**	**0.094**	**0.000**	**0.020**	**0.000**	**0.000**	**0.806**
	MM8	10/17/97 14:10	10/18/97 11:05	20.9	0.335	0.478	0.048	0.048	0.000	0.096	0.000	0.000	1.004
		10/18/97 11:00	10/19/97 10:00	23.0	0.087	0.652	0.000	0.043	0.000	0.000	0.000	0.000	0.783
			Mean	**22.0**	**0.211**	**0.565**	**0.024**	**0.046**	**0.000**	**0.048**	**0.000**	**0.000**	**0.893**
	MM9	10/16/97 13:30	10/17/97 9:45	20.3	0.247	0.198	0.099	0.000	0.000	0.049	0.000	0.000	0.593
		10/18/97 10:50	10/19/97 9:30	22.7	0.088	0.309	0.044	0.000	0.000	0.000	0.000	0.000	0.441
			Mean	**21.5**	**0.168**	**0.253**	**0.071**	**0.000**	**0.000**	**0.025**	**0.000**	**0.000**	**0.517**
	All		**Mean**	**21.8**	**0.188**	**0.352**	**0.034**	**0.056**	**0.000**	**0.032**	**0.000**	**0.000**	**0.662**
Reference	MM10	10/19/97 14:05	10/20/97 10:30	20.4	0.490	0.196	0.049	0.000	0.000	0.098	0.098	0.000	0.931
		10/21/97 11:35	10/22/97 10:15	22.7	0.618	0.176	0.132	0.044	0.000	0.000	0.176	0.000	1.147
			Mean	**21.5**	**0.554**	**0.186**	**0.091**	**0.022**	**0.000**	**0.049**	**0.137**	**0.000**	**1.039**
	MM11	10/19/97 14:20	10/20/97 11:10	20.8	0.336	0.240	0.096	0.000	0.000	0.048	0.288	0.000	1.008
		10/21/97 11:20	10/22/97 9:40	22.3	0.313	0.090	0.000	0.045	0.000	0.000	0.313	0.000	0.761
			Mean	**21.6**	**0.325**	**0.165**	**0.048**	**0.022**	**0.000**	**0.024**	**0.301**	**0.000**	**0.885**
	MM12	10/20/97 9:45	10/21/97 11:50	26.1	0.230	0.192	0.422	0.153	0.077	0.000	0.000	0.000	1.073
		10/22/97 11:20	10/23/97 9:55	22.6	0.000	0.443	0.221	0.177	0.089	0.000	0.000	0.000	0.930
			Mean	**24.3**	**0.115**	**0.317**	**0.322**	**0.165**	**0.083**	**0.000**	**0.000**	**0.000**	**1.002**
	MM13	10/20/97 10:10	10/21/97 12:30	26.3	0.304	0.038	0.000	0.000	0.000	0.000	0.076	0.000	0.418
		10/22/97 11:30	10/23/97 10:30	23.0	0.217	0.087	0.217	0.043	0.043	0.000	0.261	0.043	0.913
			Mean	**24.7**	**0.261**	**0.062**	**0.109**	**0.022**	**0.022**	**0.000**	**0.168**	**0.022**	**0.665**
	All		**Mean**	**23.0**	**0.314**	**0.183**	**0.142**	**0.058**	**0.026**	**0.018**	**0.152**	**0.005**	**0.898**

Similarly, liver size, gonad size (Figure 4.10; Table 4.9) and fecundity (numbers of eggs) (Figure 4.11) appeared greater at age for northern pike and similar for white sucker in Bell Creek relative to English River fish.

4.5.2 Fish Tissues

Tissue metal and metallothionein concentrations are provided in detail in Appendix 6. Data summaries are presented in Figures 4.12 to 4.14 for metals, Figure 4.15 for metallothionein (MT), and in Table 4.10 for both metals and MT.

Metal analyses showed some apparent reference-exposure area differences for a few metals in some tissues. Tissues accumulating the highest metal concentrations were kidney for cadmium (both species) and liver for copper (both species). Zinc was accumulated most by liver in sucker and kidney in pike. In white sucker, lead was higher in gills of exposed fish, whereas there was less difference between areas for lead in liver and kidney. In pike, lead was higher in gills and kidney in exposed fish, and there were no area differences in liver or muscle lead levels. Zinc in pike appeared more elevated in reference fish gill than in gill from exposed fish. Selenium was, on average, greater in exposed sucker and pike than in reference fish for all tissue types, and among all metals showed the most consistent reference-exposure difference. Water chemistry data show trace levels of selenium in the treated effluent (0.004 mg/L) but undetectable concentrations in all samples from the environment (<0.002 mg/L), suggesting that the observed reference-exposure difference in tissue selenium could be mine-related (refer to Appendix 3).

Recent studies have shown an ameliorative effect of tissue selenium concentrations on the bioaccumulation of mercury (Jack Klaverkamp, Freshwater Institute, pers. comm.). That is, selenium may inhibit the accumulation of mercury in fish tissues (Turner and Swick, 1983). In order to explore this relationship with the Mattabi data, plots of mercury against selenium were done for each tissue type from each species (Figure 4.14). This relationship appears to apply to muscle tissue in both white sucker and northern pike and less so in the other tissues, although the relationship was evident in white sucker liver and northern pike kidney.

As illustrated in Figure 4.15, tissue MT concentrations showed no obvious substantial reference-exposure difference for any tissue in either species. Concentrations of MT were highest in liver and lowest in gill in both species.

Table 4.9: Summary of Biological Characteristics of Northern Pike and White Sucker, Mattabi Mine (values are mean ± 1 S.E.).

Biological Measurement	Reference Areas				Exposure Areas			
	Northern Pike		White Sucker		Northern Pike		White Sucker	
	Females	Males	Females	Males	Females	Males	Females	Males
Sample Size	29	21	21	21	20	16	21	21
Mean Age (yrs)	5 ± 0.3	4 ± 0.4	10 ± 0.6	14 ± 1.0	4 ± 0.4	5 ± 0.7	12 ± 0.5	10 ± 0.4
Mean Fork Length (cm)	53.6 ± 1.59	44.1 ± 1.24	45.9 ± 0.419	44.2 ± 0.448	57.0 ± 1.96	55.7 ± 2.62	46.0 ± 1.15	43.2 ± 0.479
Mean Total Length (cm)	57.3 ± 1.66	47.2 ± 1.31	50.4 ± 0.348	48.1 ± 0.482	60.7 ± 2.07	59.5 ± 2.80	50.0 ± 1.25	46.8 ± 0.536
Mean Weight (g)	1329 ± 130	693.3 ± 65.9	1707 ± 66.5	1480 ± 43.4	1723 ± 197	1535 ± 181	1714 ± 115	1412 ± 45.5
Mean Gonad Weight (g)	22.5 ± 2.86	7.69 ± 1.31	105 ± 6.89	65.8 ± 3.02	42.3 ± 5.12	16.9 ± 2.05	92.0 ± 8.46	54.7 ± 2.21
Mean Liver Weight (g)	14.6 ± 1.64	6.27 ± 0.789	28.3 ± 1.14	17.3 ± 0.849	21.6 ± 2.25	12.7 ± 1.55	27.6 ± 2.26	18.8 ± 0.707
Mean Fecundity (eggs/fe	8998 ± 945	ot applicabl	9866 ± 119	ot applicabl	4276 ± 149	ot applicabl	0476 ± 167	ot applicable

Table 4.10: Summary of Tissue Metallothionein and Selected Metal Concentrationsµg/g fresh weight, Mattabi Mine (values are mean ± 1 S.E.).

Component	Reference Areas							
	Northern Pike				White Sucker			
	Liver	Kidney	Gill	Muscle	Liver	Kidney	Gill	Muscle
Metallothione	290 ± 34.7	99.1 ± 6.52	2.54 ± 0.256	not applicable	409 ± 59.9	105 ± 8.29	34.3 ± 2.47	not applicable
Cadmium	0.088 ± 0.015	0.269 ± 0.041	0.007 ± 0.0003	0.0195 ± 0.000	0.266 ± 0.027	1.78 ± 0.231	0.0142 ± 0.001	0.020 ± 0.0004
Cobalt	0.036 ± 0.004	0.126 ± 0.009	0.0121 ± 0.001	0.005 ± 2.12E-1	0.028 ± 0.0016	0.177 ± 0.023	0.0352 ± 0.003	0.005 ± 2.12E-11
Copper	11.9 ± 1.78	1.02 ± 0.026	0.686 ± 0.033	0.270 ± 0.016	13.7 ± 2.23	1.59 ± 0.074	0.779 ± 0.030	0.401 ± 0.027
Lead	0.232 ± 0.008	0.286 ± 0.009	0.041 ± 0.013	0.005 ± 2.12E-1	0.234 ± 0.007	0.292 ± 0.012	0.0258 ± 0.003	0.005 ± 2.12E-11
Mercury	1.25 ± 0.301	0.482 ± 0.064	0.143 ± 0.028	0.590 ± 0.053	0.142 ± 0.016	0.119 ± 0.016	0.0278 ± 0.002	0.300 ± 0.023
Selenium	1.14 ± 0.059	1.04 ± 0.048	0.126 ± 0.022	0.160 ± 0.006	0.742 ± 0.034	0.788 ± 0.037	0.076 ± 0.004	0.207 ± 0.0116
Zinc	31.7 ± 1.83	114 ± 6.06	59.4 ± 12.0	3.36 ± 0.107	27.2 ± 1.21	19.5 ± 0.567	11.8 ± 0.195	2.74 ± 0.100

Component	Exposure Areas							
	Northern Pike				White Sucker			
	Liver	Kidney	Gill	Muscle	Liver	Kidney	Gill	Muscle
Metallothione	274 ± 38.5	148 ± 11.7	3.96 ± 0.57	not applicable	386 ± 51.1	97.8 ± 5.23	23.4 ± 1.98	not applicable
Cadmium	0.093 ± 0.023	0.208 ± 0.017	0.093 ± 0.0013	0.0216 ± 0.000	0.159 ± 0.0126	1.20 ± 0.148	0.0075 ± 0.000	0.019 ± 0.0011
Cobalt	0.044 ± 0.005	0.203 ± 0.021	0.044 ± 0.009	0.005 ± 2.27E-1	0.0284 ± 0.002	0.136 ± 0.0102	0.042 ± 0.0042	0.005 ± 2.27E-11
Copper	11.2 ± 1.79	1.10 ± 0.122	11.2 ± 0.108	0.265 ± 0.014	13.8 ± 1.93	1.71 ± 0.0376	0.827 ± 0.0405	0.349 ± 0.017
Lead	0.241 ± 0.0100	0.365 ± 0.019	0.241 ± 0.051	0.005 ± 2.27E-1	0.274 ± 0.0123	0.327 ± 0.010	0.101 ± 0.014	0.005 ± 2.27E-11
Mercury	0.303 ± 0.149	0.0728 ± 0.020	0.303 ± 0.016	0.163 ± 0.040	0.0156 ± 0.002	0.0184 ± 0.003	0.0148 ± 0.002	0.059 ± 0.0094
Selenium	3.80 ± 0.213	3.48 ± 0.353	3.80 ± 0.099	1.87 ± 0.157	2.79 ± 0.147	2.99 ± 0.180	0.334 ± 0.016	1.46 ± 0.119
Zinc	27.3 ± 1.78	118 ± 11.5	27.3 ± 11.3	4.11 ± 0.139	26.3 ± 1.65	21.1 ± 0.765	13.7 ± 0.268	2.93 ± 0.123

Correlation analysis of metals in tissues versus metallothionein in tissues indicates some apparent relationships (Tables 4.11a and 4.11b). The strongest relationship occurs between copper and MT in pike liver, although the molar sum of Cd, Cu and Zn and MT in northern pike liver also shows a strong relationship. The occurrence of significant relationships within tissues, for reference and exposure areas combined, coupled with an apparent lack of area differences in most tissue metals and MT in fish (Figures 4.13 and 4.15), implies a cause-effect linkage between metals and MT but that the tissue responses were unrelated to a mine effect. All significant correlations for liver are positive in both species, whereas significant correlations for gill and kidney are usually negative. Many of the significant correlations, other than those noted above for pike liver, may be spurious.

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Table 4.11a: Correlation Matrix for Tissue Metal and Metallothionein Concentratio at Mattabi Mines

White Sucker

	MT			Probabilities (1-tailed test)			
	Gill	Kidney	Liver	Gill	Kidney	Liver	N
CdCuZn_Gill	-0.2647			0.0787			30
CdCuZn_Kidney		-0.2245			0.1165		30
CdCuZn_Liver			0.4532			0.0052	31
Aluminum_Gill	-0.1070			0.2833			31
Aluminum_Kidney		0.0510			0.3945		30
Aluminum_Liver			-0.0789			0.3366	31
Arsenic_Gill	-0.0116			0.4754			31
Arsenic_Kidney		0.1097			0.2819		30
Arsenic_Liver			0.5341			0.0010	31
Chromium_Gill	-0.1285			0.2454			31
Chromium_Kidney		0.1238			0.2573		30
Chromium_Liver			0.0368			0.4221	31
Nickel_Gill	0.0041			0.4913			31
Nickel_Kidney		-0.1474			0.2186		30
Nickel_Liver			0.0273			0.4420	31
Lead_Gill	-0.4581			0.0048			31
Lead_Kidney		-0.0040			0.4916		30
Lead_Liver			0.0471			0.4006	31
Selenium_Gill	-0.6122			0.0001			31
Selenium_Kidney		-0.1027			0.2946		30
Selenium_Liver			0.0571			0.3801	31
Zinc_Gill	-0.2707			0.0704			31
Zinc_Kidney		-0.3427			0.0319		30
Zinc_Liver			0.4905			0.0025	31

significant correlation with $\alpha = 0.05$

Note: all tissue metal concentrations are log transformed

CdCuZn = Σ Cd, Cu and Zn in µmol/gram fresh weight

Table 4.11a: Correlation Matrix for Tissue Metal and Metallothionein Concentratio at Mattabi Mines

White Sucker

	MT			Probabilities (1-tailed test)			
	Gill	Kidney	Liver	Gill	Kidney	Liver	N
CdCuZn_Gill	-0.2647			0.0787			30
CdCuZn_Kidney		-0.2245			0.1165		30
CdCuZn_Liver			**0.4532**			0.0052	31
Aluminum_Gill	-0.1070			0.2833			31
Aluminum_Kidney		0.0510			0.3945		30
Aluminum_Liver			-0.0789			0.3366	31
Arsenic_Gill	-0.0116			0.4754			31
Arsenic_Kidney		0.1097			0.2819		30
Arsenic_Liver			**0.5341**			0.0010	31
Chromium_Gill	-0.1285			0.2454			31
Chromium_Kidney		0.1238			0.2573		30
Chromium_Liver			0.0368			0.4221	31
Nickel_Gill	0.0041			0.4913			31
Nickel_Kidney		-0.1474			0.2186		30
Nickel_Liver			0.0273			0.4420	31
Lead_Gill	**-0.4581**			0.0048			31
Lead_Kidney		-0.0040			0.4916		30
Lead_Liver			0.0471			0.4006	31
Selenium_Gill	**-0.6122**			0.0001			31
Selenium_Kidney		-0.1027			0.2946		30
Selenium_Liver			0.0571			0.3801	31
Zinc_Gill	-0.2707			0.0704			31
Zinc_Kidney		**-0.3427**			0.0319		30
Zinc_Liver			**0.4905**			0.0025	31

significant correlation with $\alpha = 0.05$
Note: all tissue metal concentrations are log transformed
CdCuZn = Σ Cd, Cu and Zn in µmol/gram fresh weight

Table 4.11b: Correlation Matrix for Tissue Metal and Metallothionein Concentrati at Mattabi Mines

Northern Pike

	MT			Probabilities (1-tailed test)			
	Gill	Kidney	Liver	Gill	Kidney	Liver	N
CdCuZn_Gill	**-0.4094**			0.0123			30
CdCuZn_Kidney		-0.2052			0.1341		31
CdCuZn_Liver			**0.8028**			2.76E-08	31
Aluminum_Gill	0.2815			0.0625			31
Aluminum_Kidney		**-0.5593**			0.0005		31
Aluminum_Liver			**0.3778**			0.0181	31
Arsenic_Kidney		**-0.3487**			0.0474		24
Arsenic_Liver			-0.0241			0.4487	31
Cadmium_Gill	0.2616			0.0776			31
Cadmium_Kidney		**-0.4168**			0.0098		31
Cadmium_Liver			**0.4956**			0.0023	31
Chromium_Gill	0.0529			0.3888			31
Chromium_Kidney		-0.1682			0.1829		31
Chromium_Liver			0.2641			0.0756	31
Copper_Gill	-0.1410			0.2247			31
Copper_Kidney		0.1358			0.2333		31
Copper_Liver			**0.9060**			5.43E-11	31
Iron_Gill	**0.3624**			0.0225			31
Iron_Kidney		-0.1864			0.1577		31
Iron_Liver			**0.3988**			0.0131	31
Mercury_Gill	**-0.3444**			0.0289			31
Mercury_Kidney		**-0.4242**			0.0087		31
Mercury_Liver			**0.4454**			0.0060	31
Nickel_Gill	-0.1674			0.1840			31
Nickel_Kidney		-0.1797			0.1667		31
Nickel_Liver			0.2085			0.1302	31
Lead_Gill	-0.0167			0.4645			31
Lead_Kidney		**0.4533**			0.0052		31
Lead_Liver			0.0929			0.3096	31
Selenium_Gill	0.2522			0.0855			31
Selenium_Kidney		0.2460			0.1233		24
Selenium_Liver			-0.0043			0.4909	31
Zinc_Gill	**-0.3690**			0.0205			31
Zinc_Kidney		-0.2052			0.1341		31
Zinc_Liver			**0.5581**			0.0006	31

significant correlation with $\alpha = 0.05$

Note: all tissue metal concentrations are log transformed

CdCuZn = Σ Cd, Cu and Zn in µmol/gram fresh weight

Table 4.11b: Correlation Matrix for Tissue Metal and Metallothionein Concentrati at Mattabi Mines

Northern Pike

	MT			Probabilities (1-tailed test)			
	Gill	Kidney	Liver	Gill	Kidney	Liver	N
CdCuZn_Gill	[illegible]			0.0123			30
CdCuZn_Kidney		-0.2052			0.1341		31
CdCuZn_Liver			[illegible]			2.76E-08	31
Aluminum_Gill	0.2815			0.0625			31
Aluminum_Kidney		[illegible]			0.0005		31
Aluminum_Liver			[illegible]			0.0181	31
Arsenic_Kidney		[illegible]			0.0474		24
Arsenic_Liver			-0.0241			0.4487	31
Cadmium_Gill	0.2616			0.0776			31
Cadmium_Kidney		[illegible]			0.0098		31
Cadmium_Liver			[illegible]			0.0023	31
Chromium_Gill	0.0529			0.3888			31
Chromium_Kidney		-0.1682			0.1829		31
Chromium_Liver			0.2641			0.0756	31
Copper_Gill	-0.1410			0.2247			31
Copper_Kidney		0.1358			0.2333		31
Copper_Liver			[illegible]			5.43E-11	31
Iron_Gill	[illegible]			0.0226			31
Iron_Kidney		-0.1864			0.1577		31
Iron_Liver			[illegible]			0.0131	31
Mercury_Gill	[illegible]			0.0289			31
Mercury_Kidney		[illegible]			0.0087		31
Mercury_Liver			[illegible]			0.0060	31
Nickel_Gill	-0.1674			0.1840			31
Nickel_Kidney		-0.1797			0.1667		31
Nickel_Liver			0.2085			0.1302	31
Lead_Gill	-0.0167			0.4645			31
Lead_Kidney		[illegible]			0.0052		31
Lead_Liver			0.0929			0.3096	31
Selenium_Gill	0.2522			0.0855			31
Selenium_Kidney		0.2460			0.1233		24
Selenium_Liver			-0.0043			0.4909	31
Zinc_Gill	[illegible]			0.0205			31
Zinc_Kidney		-0.2052			0.1341		31
Zinc_Liver			[illegible]			0.0006	31

[illegible] significant correlation with $\alpha = 0.05$

Note: all tissue metal concentrations are log transformed

CdCuZn = Σ Cd, Cu and Zn in µmol/gram fresh weight

5.0 HYPOTHESIS TESTING

5.1 Methods

The 12 hypotheses tested at Mattabi are listed in Table 5.1, along with a more specific listing of the "effect" (response) and "exposure" (predictor) variables to be examined under each hypothesis. The general reasoning behind all of these hypotheses is that a mine "effect" is a measurable difference between reference and exposure locations, and/or a trend between locations that are exposed to different degrees of contamination. The hypotheses address either the ability of a particular monitoring tool to detect such an effect (and, in aggregate, whether an effect exists), or the **relative** ability of two different monitoring tools, that are being compared to one another, to detect such an effect. H5 through H8 are of the first type, whereas H1 through H4 are of the second type. H9 through H12 address the **relative** ability of two monitoring tools to detect a correlation between specific predictor and response variables.

These different types of hypotheses require different methods of statistical analysis. The following subsections describe the statistical approach in each category. In all cases, appropriate data transformations were applied prior to statistical analysis, such as log transformation for chemical concentrations or other parameters that span a wide range, and arcsine square root transformations for percent response variables. A significance criterion was used for all the statistical analyses, and use of the term 'significant" implies that this criterion was met.

It should be recognized that the term "predictor" variable is not intended to mean that the measure of exposure used (e.g., metal concentration in water) can be used to "predict" a specific biological response at all mine sites or in other surveys at this mine site. Nor does it imply that the predictor is necessarily the cause of a biological effect. Rather, the predictive ability is only suggested by correlation between effect and exposure measures.

5.1.1 H1 - Comparison of Sediment Toxicity Tests

Hypothesis H1 addresses the **relative** ability of three sediment toxicity test tools (response measures) to detect a mine effect. In particular, the *Hyalella azteca*, *Chironomus riparius* and *Tubifex tubifex* tests were compared to determine whether these tools differ in their ability to detect a mine effect (i.e., a reference versus exposure area difference, or a trend with degree of exposure within the exposure area). An area identifier, ordered within the exposure area to reflect

TABLE 5.1 VARIABLES AND HYPOTHESES ATMATTABI

Hypothesis	Y variables	X variables	Test (Ho)	Comment
H1	Sediment Toxicity Response i Sediment Toxicity Response j	Lake Area Identifier	no trend or area x tool interaction by ANOVA	*Hyalella, Chironomus* and *Tubifex* tests are the monitoring tools of interest.
H2	Metal i in Tissue i Metal i in Tissue j	Creek Identifier	no R/E tool interaction	Tissues for white sucker and northern pike done separately by sex, as required.
H3	MT in Tissue i MT in Tissue j	Creek Identifier	no R/E tool interaction	Tissues for white sucker and northern pike done separately by sex, as required.
H4	Metal i in Tissue j MT in Tissue j	Creek Identifier	no R/E tool interaction	Tissues for white sucker and northern pike done separately by sex, as required.
H5	CPUE for sucker, pike and all fish	Creek Identifier	no R/E difference by ANOVA	CPUE by species and for all fish in Bell Creek and reference creek using gill net.
H6	BPUE, No. of Fish Taxa	Creek Identifier	no R/E difference by ANOVA	CPUE by species and for all fish in Bell Creek and reference creek using gill net.
H6 (benthos)	No. of Taxa Benthic Density Indicator Taxa	Area Number in Order of Decreasing Sediment Metal Concentration	no trend or R/E difference by ANOVA	Collections at 3 stations per area, 5 exposure areas and 2 reference areas.
H7	Length at age Weight at age Weight at length	Creek Identifier	no R/E difference by ANOVA	Analysis done separately for males and females (pike and sucker). Used age as a covariate as appropriate.
H8	Liver weight, gonad weight by sex, at age. Fecundity at age (females).	Creek Identifier	no R/E difference by ANOVA	Mature white sucker and northern pike.
H9*	No. of Fish Taxa Length and Weight at age Gonad and Liver wt at age and weight Fecundity	Dissolved Metal in Water (Tool 1) Total Metal in Water (Tool 2)	same correlation	Water quality in Bell Creek reflects mine influence.
H10	Benthic Density No. of Benthic Taxa Indicator Taxa Sediment Toxicity Endpoints	Partial Metal i in Sediment (1) Total Metal i in Sediment (2) SEM/AVS Ratio	same correlation	Use various sediment chemistry results.
H11	Benthic Density No. of Benthic Taxa Indicator Taxa	Sediment Toxicity Results	same correlation	Use various toxicity endpoints (*Hyalella, Chironomus*, *Tubifex* tests).
H12*	Metal i in Tissue j MT in Tissue j	Metal i in Water (dissolved and total)	same correlations	Tissues for white sucker and northern pike done separately by sex, as required.
Sediment Triad Hypotheses	Benthic PCs Sediment Toxicity Endpoints Sediment Chemistry PCs	Benthic Variables (B) Toxicity Variables (T) Chemistry Variables (C)	no correlation C-B, C-T and B-T	Sphericity test Mantel's test

Definitions:
MT = metallothionein
R/E = reference/exposure
CPUE = catch-per-unit-effort (number of fish caught per unit fishing effort)
BPUE = biomass-per-unit-effort (mass of fish caught per unit fishing effort)

* H9 and H12 are C/I comparisons with reference stations included in the correlations. Results to be interpreted cautiously.

distance from the mine site (i.e., MME1 to MME5), was used as a surrogate for degree of exposure to mine-related contaminants, based on the fact that with increased distance there is an attenuation in contaminant levels. The use of direct measures of exposure in evaluating sediment toxicity test results is included within the context of the overall Sediment Quality Triad hypothesis (Section 2.3.5). Analysis of variance (ANOVA) was used to address this hypothesis, as described below.

Essentially, the ANOVA is used to compare tool effectiveness in two ways:

- by determining if there is a reference area - exposure area difference in mean values for each tool (a larger difference indicates greater effectiveness); and
- by determining if there is a linear trend or gradient in response within the exposure area (a significant trend and greater slope indicates greater effectiveness).

The ANOVA partitions overall variance in the response measure into a number of terms, representing effects of particular interest. These include:

- A "Ref vs Exp x Tool" term which indicates whether the Reference versus Exposure difference is similar for both tools being compared (e.g., for *Hyalella* toxicity and *Tubifex* toxicity). It measures how much the spread between Line 1 and Line 2 differs from the spread between Line 3 and Line 4 in Figure 5.1. Lines 1 to 4 represent the means of the response measures for each tool in the reference or exposure area. This term also indicates how much the Line 1 to Line 3 spread differs from the Line 2 to Line 4 spread, or the degree of difference between the slopes of the two lines shown in Figure 5.2. A larger difference between the reference and exposure means for one tool relative to the other would indicate a greater effectiveness for the tool with the greater difference. For this example, the absolute reference-exposure difference for each tool is small, but the differences are in opposite directions. This produces a significant Ref vs Exp x Tool interaction, which implies that Tool 1 (*Hyalella* growth) is more effective than Tool 2 (*Tubifex* reproduction). The interaction is also illustrated in Figure 5.2.

- A "Linear Trend x Tool" term which indicates whether the linear trend in the Exposure area (e.g., from near-field to far-field) is similar for both tools. It

measures how much the Line 2 to Line 5 spread differs from the Line 4 to Line 6 spread in Figure 5.1. This term also indicates the degree of difference between the Line 5 and Line 6 slopes. A greater slope in the Line 5 (Tool 1) than in Line 6 (Tool 2) indicates a greater effectiveness of Tool 1 in this example.

In all cases, to test whether the spread described in either of the above two "effect" terms is significant, each is compared to the spread of the exposure means for each reach around Lines 5 and 6 (i.e., to a lack of fit "error" term). If the "effect" variance is large relative to the "error" variance, then the effect is considered to be present, and the tool is concluded to be responsive to mine exposure.

The "lack of fit" spread is compared in turn to the overall "within reach" spread (i.e., between stations in a particular reach), in order to test whether there may be any other (i.e., non-linear) trend among the exposure means, that is whether a straight line can be drawn through response measures for all exposure reaches. If "lack of fit" is significant, the nature of the trend is examined and, if appropriate, the analysis is repeated using a non-linear (second order) trend term instead of a linear trend term. This would appear in Figure 5.1 as curved lines rather than the straight Lines 5 and 6.

The response measures for H1 (*Hyalella* or *Tubifex* toxicity) were standardized prior to statistical analysis, in order to make them equally variable within a reach, since homogeneity of variance is an assumption of the ANOVA procedure. The standardization procedure involves dividing the *Hyalella* growth values by the pooled within-reach standard deviation for *Hyalella* growth, and dividing the *Tubifex* young production values by the pooled within-reach standard deviation for *Tubifex* production of young.

5.1.2 H2 through H4 - Fish Tissue Metals and Metallothionein

These hypotheses also test the relative ability of related exposure tools in fish to detect a mine effect. However, unlike Hypothesis H1, for Hypotheses H2 to H4 there is only a single level of exposure and mine effects are identified only by detection of reference-exposure differences using ANOVA. A test of "trend" is simply by comparison of responses at the reference and exposure areas. A significant interaction between the two tools being compared suggests a greater effectiveness in the tool with the larger difference between exposure area response and reference area response. Figure 5.3 illustrates this approach.

5.1.3 H5 through H8 - Fish Communityand Fish Health

These hypotheses test the ability of individual monitoring tools to detect a mine effect (fish catch-per-unit-effort, fish growth, etc.). To determine if a fish monitoring index can detect a mine effect, a simple ANOVA test is used to determine whether the index varies more between areas than it does within areas. If so, then the pattern of differences between areas is examined to confirm that the response is consistent with a mine effect.

Figure 5.4 illustrates the approach. The patterns on the top graph are consistent with a mine effect (decreased fish catch near the mine). The bottom graph illustrates effects not typically consistent with a deleterious mine effect. However, judgement is always needed in interpretation of response patterns. For example, an increased fish catch near the mine could represent a mine effect if the mine caused nutrient enrichment rather than toxicity.

For both northern pike and sucker, fish growth and organ size were examined at age because the response measures can vary with fish age. Therefore, an age covariate was added to the ANOVA model to adjust all fish to a common age. To address the situation where fish grow at different rates in different areas, the analysis was also carried out with body weight as a covariate (Environment Canada, 1998). In addition, growth and organ size differed by sex for both species. Therefore, all analyses for H7 and H8 were carried out by sex.

5.1.4 H6 - Benthic Community Structure

Hypothesis H6, when considered with respect to the No Name Lake-Mine Creek Bay benthic communities, was tested across reference and exposure areas to assess reference-exposure differences, and within the exposure area for trends within the gradient. An area identifier, ordered within the exposure area to reflect relative position within the sediment quality gradient (E1 = highest metal concentrations, E5 = lowest concentrations), was used as a measure of mine exposure. ANOVA was used to address this hypothesis, as described below.

The ANOVA partitions overall variance in the response measure into a number of terms representing effects of particular interest. These include:

- An "Among Reference" term which indicates whether the Reference reaches are similar to each other. It measures the spread of reference means around Line 1 in Figure 5.5 (i.e., around the grand reference mean represented by the solid line).

This term is quantified in order to indicate whether reference reaches are differentially influenced by some factor that may also be confounding effects in the exposure area.

- A "Ref vs Exp" term which indicates whether the Reference and Exposure reaches are similar to each other. It measures the spread between Line 1 (reference mean) and Line 2 (exposure mean) in Figure 5.5 (i.e., between reference and exposure means). A reference-exposure difference is generally indicative of tool effectiveness, assuming that the direction of the difference is consistent with impact.
- A "Linear Trend" term which indicates whether there is a linear trend in the Exposure area (e.g., from near-field to far-field). It measures the spread between Line 2 and Line 3 (the exposure trend line) in Figure 5.5 (i.e., the difference in slopes). A significant linear trend, i.e., a near-field to far-field gradient is indicative of tool effectiveness, assuming that its direction is consistent with impact.

In all cases, to test whether the spread is significant, as described in any of the above three "effect" terms, each is compared to the spread of exposure reach means around Line 3 (i.e., to a "lack of fit" error term). This "lack of fit" error term accounts for the residual variability in the data after the above three terms are subtracted from the total among-reach variability. If an "effect" term is large relative to the "lack of fit" error, then the effect is more likely to be significant.

The "lack of fit" spread is compared in turn to the overall "within reach" spread (i.e., between stations within a reach), in order to test whether there may be any other (i.e., non-linear) trend among the exposure means, that is whether a straight line is the best description of the trend. If "lack of fit" is significant, the nature of the trend is examined and, if appropriate, the analysis is repeated using a non-linear (second order) trend term instead of a linear trend term. This would appear in Figure 5.4 as a curved line rather than straight Line 3.

In the example, the data points in Figure 5.5 represent numbers of benthic taxa in each area and station. The ANOVA shows a significant "Ref vs Exp" effect, because there is a substantial difference between Lines 1 and 2. The ANOVA also shows that there is a significant "Linear Trend" effect, because numbers of taxa are lowest near the mine (Reach

E1) and increase as we move further away (i.e., slope of Line 3). The interpretation would be that benthic species richness is responding to mine exposure.

5.1.5 H9 through H12 - Tool Integration Hypotheses

Hypotheses H9 to H12 address the **relative** ability of two monitoring tools to detect a mine effect. For example, in H10, partial metal in sediment was compared to total metal in sediment, for each of the key metals, to determine whether these two monitoring tools differ in their ability to detect a mine effect (i.e., a correlation between a biological response measure, such as fish CPUE, and the metal predictor variable). Correlation analysis was used to address this hypothesis, as described below.

The squared coefficient of correlation (r^2) between the response measure (Y) and each predictor variable (X1 or X2) indicates the proportion of variance in the response measure that is explained by the predictor (Figure 5.6). The best predictor, for each pair compared, is the one which explains the highest proportion of variance (i.e., has the highest r). No statistical test was performed to determine whether r_1 differs significantly from r_2, because the two r values are based on the same Y data set and are not independent. However, the individual r values were tested for statistical significance. Two r values were compared, to draw inferences about which monitoring tool is better, only when at least one of the r values was of the correct sign (negative or positive) to suggest a mine effect, and statistically distinguishable from zero based on a one-tailed test.

When differences between r values are small (e.g., ≤ 0.1), even though one or both r values may be statistically significant, a judgement is generally not made that the tool with the slightly higher r value is better able to detect an effect. Also, the correlations are generally calculated for many exposure measures (metals), so that judgements with respect to which exposure measure tool (e.g., total versus dissolved metal concentration in water) is more strongly correlated with biological response are made by the weight-of-evidence based on all r values for each tool. The exposure and response measures selected for inclusion in this analysis were those which showed an apparent spatial relationship to the mine site (i.e., trend among exposure reaches or difference between reference and exposure reaches).

These correlations can be computed in two ways - including and excluding reference stations. Response tools correlated with causal agents when reference sites are excluded are considered more effective than those showing correlations when reference sites are included.

This is because correlations seen within the exposure gradient are clearly associated with mine impact. The inclusion of data from the two references areas could potentially impose spurious correlations by producing clusters of data points at low exposure concentrations.

For H9 and H12, which involve correlations between responses in fish and water quality, samples were collected from four exposure stations within a larger exposure area and at another four stations in a reference area. Aqueous metal concentrations in the exposure area showed some variation among stations, with the principal difference occurring between MME6 and the other stations owing to effluent mixing patterns. Thus, the study design for these hypotheses can be considered either a control/impact (C/I) design or a control/impact/impact (C/I/I) design, depending on whether large fish range throughout the exposure area on the time scale of fish response and fish movement. For example, MT concentrations in fish could conceivably respond relatively quickly to exposure concentrations (as observed at Heath Steele, BEAK, 1998), although a measurable growth response to a specific exposure level in large fish would probably require that fish be relatively sedentary and not range throughout the exposure area at Mattabi. To address the possibility that fish are able to respond to exposure conditions at a relatively local level, H9 and H12 are tested by correlation analysis. In so doing, it is recognized that the results of the analyses must be interpreted with caution because the large fish sampled are likely to range freely throughout the exposure area and to be exposed to spatially averaged conditions. Also, reference station data are included in the analysis to offset the very limited variation in exposure conditions measured at exposure stations.

For H10 and H11, which are sediment-related hypotheses, correlation analysis can be reasonably completed using only data from the five exposure areas sampled. Thus, H10 and H11 are tested without the use of reference site data.

Hypothesis H9

At Mattabi, H9 (relationship between water chemistry and biological variables) is tested both using fish health and community tools. This hypothesis compares the effectiveness of dissolved versus total metals in water as predictors of biological response. As noted above, this analysis is of limited value because it cannot reasonably be done exclusive of reference stations and because of the limitations imposed by the C/I design. Some of the biological

responses observed in H5 through H8 did not appear to be in response to metal exposure (e.g., greater growth, organ size, etc.). However, because the underlying nature of dose-response patterns in the natural environment is uncertain, H9 has been tested under the general assumption that the response could be *potentially* attributed either directly or indirectly to mining effects.

Hypothesis H9 is not tested using benthic community tools. This is because most of the exposure areas for benthos are located in Mine Creek Bay, where water-borne metal concentrations are greatly diluted by Sturgeon Lake. Water quality variations from station to station here are small and concentrations low relative to Canadian surface water guidelines. Thus, it is unlikely that benthic effects would be attributed to water quality, with the possible exception of effects in No Name Lake (area HE1).

Hypothesis H10

Hypothesis H10 tests both benthic index versus sediment chemistry correlations and sediment toxicity versus sediment chemistry correlations. The sediment chemistry tools include total metal concentrations (hydrogen peroxide/nitric acid extraction), partial metal concentrations (hydroxylamine extraction) and the ratio of the molar sum of simultaneously extracted metals (SEM) and acid volatile sulphide (AVS). Metals included in the SEM value are Cd, Cu, Ni, Pb and Zn. These are the metals often contributing to toxicity and potentially rendered non-bioavailable by the formation of metal monosulphides.

Hypothesis H11

Hypothesis H11 examines the remaining component of the "sediment quality triad" - the correlation between benthic indices and sediment toxicity. The toxicity tests include amphipod (*Hyalella azteca*), chironomid (*Chironomus riparius*) and oligochaete (*Tubifex tubifex*) tests on sediment samples from each lake station.

Hypothesis H12

H12 (relationship between water and fish tissue chemistry response) is tested using total metal concentration in water, and metallothionein and metals in fish tissues. As noted previously, H12 cannot be rigorously tested due to limitations imposed by the C/I design.

5.1.6 Triad Hypotheses

The "triad" hypothesis addresses the issue of whether chemical contaminants may be responsible for biological "effects" that are apparent in the study area. This hypothesis has not been articulated explicitly in the set of 13 hypotheses that were developed by the AETE (Section 1.0); however, it is consistent with the interest in H9 through H12 about the ability or relative ability of monitoring tools to detect correlations or relationships between chemical, toxicological and biological parameters. The basic approach to evaluation of the triad hypothesis is to simultaneously examine three types of correlations: chemical-toxicological (C-T), toxicological-biological (T-B) and chemical-biological (C-B). These are the three "arms" of the triad that would support an interpretation that chemical contaminants are responsible for biological effects. There should be significant correlations on all three arms before the hypothesis that chemical contaminants are the cause of the effect is accepted.

Statistical approaches to triad evaluation follow Green and Montagna (1996) and Chapman (1996). One approach is to examine the three bivariate correlations (C-T, T-B, C-B) for different sets of chemistry, toxicity and biology monitoring tools. Then, the overall evaluation of the triad hypothesis is based on "weight-of-evidence" considerations (i.e., are there sets of parameters showing significant C-T, T-B **and** C-B correlations, how many sets are there that meet this criterion, and how strong are the correlations in general?). This approach is simple, but rather tedious when there are many different chemistry, toxicity and biology monitoring tools to be paired in different ways.

A more holistic approach was applied using principal components analysis (PCA) to reduce the large number of variables to one or two dominant principal components (PCs) representing the mine effect gradient in chemistry (based on the original chemical variables), one or two representing the gradient in toxicity, and one or two representing the gradient in biology. Then multiple correlation coefficients (R) are computed using the PC variables to represent the dominant C-T, T-B and C-B correlations (if any) on each arm of the triad. Mantel's test was used to produce a single measure of concordance on each arm of the triad, equivalent to R^2 (e.g., Figure 5.7). Finally, Bartlett's test of sphericity was applied to determine if there is a significant overall concordance across the three arms of the triad.

5.2 Results

The general conclusions with respect to hypothesis testing at Mattabi are summarized in Table 5.2. The following sections present the findings in more detail, based on the statistical tables and figures presented in Appendix 7. The discussion is focused on results that meet the significance criterion of $p \leq 0.05$. Use of the term "significant" implies that this criterion was met, although "suggested" results may be mentioned as such when the criterion is approached but not achieved. The reader is reminded that tool effectiveness discussed herein pertains to the specific Mattabi dataset produced in this study, and conclusions should not necessarily be considered generally applicable at other mine sites.

5.2.1 H1 - Sediment Toxicity as a Response to Exposure

None of the *Tubifex, Hyalella* or *Chironomus* lethal or sublethal responses (arcsine square root of %) showed a trend that can be related to a mining effect (reference-exposure difference or exposure area trend). Significant differences in responses for *Chironomus* growth and *Tubifex* production of cocoons and young were evident among areas, but these differences did not occur between reference and exposure area groups. The lack of toxicity response to mine exposure is perhaps surprising, given the extreme sediment metal concentrations present. The small partial metals fraction may reflect this low bioavailability and low toxicity associated with sediment metals.

5.2.2 H2 - Comparison of Fish Tissues for Metal Concentration

This hypothesis was tested metal by metal in a two step process. First, ANOVA was used to identify whether a reference-exposure difference exists for each tissue (liver, gill, kidney, muscle) and metal. Effects of age and sex on metal concentration were also evaluated at this stage. Second, tissues were compared in a pair-wise fashion to identify significant area x tissue (tool) interaction, which would indicate a greater degree of response (i.e., effectiveness) in one of the two tissues. This second comparison was made only if two or more tissues showed a significantly greater concentration in the exposure area for any metal.

In female white sucker, for example, the levels of selenium in kidney and liver exhibited similar relative degrees of difference between the reference and exposure areas, as indicated by the reach*tool term which was not statistically different ($p = 0.660$; see Hypothesis H2 in Appendix 7). However, for selenium in kidney and muscle in female white sucker, there was a significant reach*tool term ($p = 5.41E\text{-}04$), indicating that the relative differences between the reference and exposure for these two tissues were not similar. To identify which tissue demonstrated the larger

TABLE 5.2: SUMMARY AND GENERAL CONCLUSIONS OF HYPOTHESES TESTED AT MATTABI

Hypothesis	Y variables	X variables	Test (Ho)	Comment
H1	Sediment Toxicity Response i Sediment Toxicity Response j	Lake Area Identifier	no trend or area x tool interaction by ANOVA	All tests showed no significant reference-exposure differences or trends in the exposure area. Thus, no discernible difference in effectiveness. Area effects unrelated to exposure were evident for sublethal responses in *Chironomus* and *Tubifex*.
H2	Metal i in Tissue i Metal i in Tissue j	Creek Identifier	no R/E x tool interaction	White sucker tissue concentrations of Pb and Zn showed exposure area elevation only in gill. Se was higher in exposed sucker muscle and kidney relative to reference suckers. These metals were also elevated in other tissues of exposed sucker, but only for one sex. Northern pike Cd and Zn in muscle, Pb and Co in gill/kidney, and Se in muscle, gill and liver showed exposure area elevation. The most effective tissue for monitoring of tissue metals depends on metal. Linkage of Se and Co to Mattabi is weak.
H3	MT in Tissue i MT in Tissue j	Creek Identifier	no R/E x tool interaction	MT response only significant for pike gill and kidney. These tissues were equally effective.
H4	Metal i in Tissue j MT in Tissue j	Creek Identifier	no R/E x tool interaction	MT and Pb, Se and Co elevated in northern pike gill/ kidney. MT and some metals (Pb, Co) were equally effective in showing mine-related exposure. Se was more elevated in the exposure area than MT for pike gill.
H5	CPUE for sucker, pike and all fish	Creek Identifier	no R/E difference by ANOVA	No significant effect of mine exposure on fish abundance.
H6	BPUE, No. of Fish Taxa	Creek Identifier	no R/E difference by ANOVA	Significant decrease in number of fish species in exposure area, probably due to habitat factors. No significant effect of mine exposure on fish biomass.
H6 (benthos)	No. of Taxa Benthic Density Indicator Taxa	Area Number in Order of Decreasing Sediment Metal Concentration	no trend or R/E difference by ANOVA	Significant decrease in exposure area, and exposure area trends in density, number of taxa and indicator taxa.
H7	Weight at age Length at age	Creek Identifier	no R/E difference by ANOVA	No significant differences in growth of white sucker. Significantly larger pike in exposure area. "Effects" of exposure beneficial and probably related to habitat.
H8	Liver weight, gonad weight by sex, at age. Fecundity at age (females).	Creek Identifier	no R/E difference by ANOVA	White sucker liver significantly larger in exposure fish. Gonad weight (body-weight adjusted) slightly smaller in exposed male sucker. Liver and gonad weight and fecundity in pike all significantly higher in exposed fish. "Effects" of exposure in pike not consistent with adverse impact.
H9*	No. of Fish Taxa Length and Weight at age Gonad and Liver wt at age (pike), Fecundity	Dissolved Metal in Water (Tool 1) Total Metal in Water (Tool 2)	same correlation	Similar correlations for dissolved and total metals. Better correlations with total Pb versus dissolved. H9 to be interpreted with caution due to study design limitations.

TABLE 5.2: SUMMARY AND GENERAL CONCLUSIONS OF HYPOTHESES TESTED AT MATTABI

Hypothesis	Y variables	X variables	Test (Ho)	Comment
H10	Benthic Density No. of Benthic Taxa Indicator Taxa Sediment Toxicity Endpoints	Partial Metal i in Sediment (1) Total Metal i in Sediment (2) SEM/AVS Ratio	same correlation	Similar correlations for total and partial metals with benthic community effects and sediment toxicity results (*Chironomus* and *Tubifex* sublethal endpoints). Total metals slightly better correlated than partial metals. No correlation of SEM/AVS ratios with benthos or toxicity results. SEM/AVS ratio did not correlate with observed toxicity results.
H11	Benthic Density No. of Benthic Taxa Indicator Taxa	Sediment Toxicity Results	same correlation	*Tubifex* reproduction showed strongest correlations with benthic metrics supporting cause-effect linkages. *Chironomus* growth showed some linkage with benthos but the direction of the correlation is inconsistent with impact.
H12*	Metal i in Tissue j MT in Tissue j	Metal i in Water (dissolved and total)	same correlations	In sucker, only Pb and Zn in gill were correlated with aqueous metal concentrations. MT concentrations in sucker tissues were unrelated to exposure concentrations. In pike, Pb in kidney and gill, and Zn in muscle, were correlated with aqueous concentrations of the same metals. Pike MT levels in kidney and gill were correlated with Cd, Pb and Zn in water. These metals in tissue showed similar correlations with corresponding metals in water.
Sediment Triad Hypotheses	Benthic PCs Sediment Toxicity Endpoints Sediment Chemistry PCs	Benthic Variables (B) Toxicity Variables (T) Chemistry Variables (C)	no correlation C-B, C-T and B-T	The sediment quality triad was significant using either partial or total metals, although the benthos/chemistry correlation was stronger using partial metals. The chemistry/toxicity and benthos/toxicity arms of the triad were not significant.

* H9 and H12 are C/I comparisons with reference stations included in the correlations. Results to be interpreted cautiously.

gradient of difference, the plot provided (following the ANOVAs for H2 in Appendix 7) indicates a greater slope for the selenium in liver relative to selenium in muscle.

Also in Appendix 7 are tables identifying the cases where reference-exposure differences occurred for each metal, and showing the directions of the differences (whether exposure or reference fish were higher in concentration). On balance, for Cd, Co, Se, Ni, Zn, Pb, most (but not all) reference-exposure differences showed higher concentrations in the exposure area in at least one tissue or species. This provides a degree of confidence that the effects observed are mining-related.

White Sucker

For white sucker, significantly higher concentrations in exposure fish were found for Pb and Zn in gill, and for Se. The Se elevation was greater in muscle than in kidney for female sucker, but greater in kidney than in muscle for male suckers. In addition, there were some exposure area elevations observed in only one sex, e.g., Pb in female livers, Se in male gills. Gill displays a greater elevation of Pb and Zn in the exposure area than does muscle, liver or kidney tissue.

Kidneys display a greater elevation of Se in the exposure area than do muscle tissues for female white sucker; however, for males, there is a greater elevation in the muscle. Livers display a greater elevation of Se in the exposure area than do muscle tissues for female white sucker. Muscle displays a greater elevation of Se in the exposure area than does gill tissue for male white sucker.

Northern Pike

In northern pike, significantly higher concentrations were found in the exposure area for Cd and Zn in muscle, Pb and Co in gill and kidney, and Se in muscle, gill and liver. In addition, there were some exposure area elevations that were observed in only one sex, e.g., Se in female kidneys, Cr in female muscle and in male kidneys. Water chemistry results do not suggest a Cr source from Mattabi.

Muscle displays a greater elevation of Cd and Zn in the exposure area than does gill, liver or kidney tissue. Gill and kidney display a greater elevation of Pb and Co in the exposure area than do liver or muscle. Gill tissue displays a greater elevation of Fe in the exposure area than does muscle, liver or kidney.

Muscle displays a greater elevation of Se in the exposure area than does gill tissue, and gill tissues display a greater elevation than liver.

5.2.3 H3 - Comparison of Fish Tissues for Metallothionein Concentration

This hypothesis was tested by identifying whether a reference-exposure difference exists for each tissue (liver, kidney, gill). Effects of age and sex on MT concentration were also evaluated at this stage.

As shown in Appendix 7, the only significant MT response occurred in gill and kidney of northern pike (both sexes combined). There was no significant interaction between tissue (gill and kidney) or exposure effects on MT response, indicating similar responses for these two tissues. However, higher levels of MT were measured in the kidney making it a more reliable tool compared to the very low levels measured in gill.

5.2.4 H4 - Metallothionein vs Metal in Fish Tissues as a Response to Exposure

The only fish tissues showing both MT and metal responses were northern pike gill and kidney. Northern pike gill and kidney showed elevated MT, Pb, Se and Co in the exposure area relative to the reference area. ANOVAs for gill and for kidney showed no significant interaction between tool (MT, metal) and exposure effects, for either Pb or Co, indicating similar effectiveness of MT and metal. However, Se showed a stronger elevation in the exposure area than MT for pike gills.

5.2.5 H5, H6 - Fish CPUE, Fish Community as Responses to Exposure

Fish CPUE

The ANOVAs for CPUE for individual species and for all fish (numbers of individuals) showed no significant difference in CPUE between reference and exposure areas. The white sucker CPUE difference was insignificant ($p = 0.072$), because of only a slightly greater abundance of sucker in the exposure area (Bell Creek) catches (Appendix 7). These results indicate no effect of mine exposure on fish abundance.

Fish BPUE

ANOVAs of biomass-per-unit-effort (BPUE) for individual fish taxa and for all fish showed no differences between areas (Appendix 7). As stated for CPUE, these results indicate no effect of mine exposure on fish abundance.

Fish Species

The ANOVA for numbers of fish species present indicates a significant reference-exposure area difference (Appendix 7). The differences are related to the presence of sauger, lake whitefish and lake herring in the reference area but not in the exposure area. This fish community difference can probably be related to the presence of accessible deepwater habitat near sampling areas in the English River, rather than to any mining-related effect.

5.2.6 H6 - Benthic Community Measures as Responses to Exposure

ANOVAs demonstrated several reference exposure area differences and/or exposure area trends in the benthic community which are consistent with mining-related effects (Appendix 7). Reference-exposure area differences were found for log total density and log (and arcsine square root %) *Chironomus* (midge) density, whereas exposure area trends were significant for log total abundance, % Hydracarina (water mite), *Pisidium* (pea clam) abundance and numbers of benthic taxa. Nearly significant ($p = 0.053$) trends were found for arcsine square root % *Chironomus* (exposure area trend) and log Hydracarina abundance (exposure area trend, $p = 0.051$; and reference-exposure difference, $p = 0.054$). The directions of these differences were consistent with mining-related effects (e.g., reduced taxa, total abundance, *Pisidium* and Hydracarina abundances; increased *Chironomus* abundance). As outlined in Section 3.1, other benthic effects were apparent but not tested, including *Psectrocladius* abundance which was extreme at MME1, with none present elsewhere (rendering statistical testing difficult but unnecessary). As discussed previously (Section 3.1.4), benthic community responses appear more likely associated with mine-related effects than with variations in benthic habitat among areas.

5.2.7 H7 - Fish Growth and Condition as a Response to Exposure

Figure 4.9 shows size-age and weight-length relationships for white sucker and northern pike. The ANOVA tables showing trends in age-adjusted lengths and weights and length-adjusted weights are presented in Appendix 7.

White Sucker

White sucker differ by sex in their growth characteristics, with males being generally smaller than females. ANCOVAs are provided for both males and females using age adjusted data to account for the effects of age on fish size (Appendix 7). The ANOVAs show no significant growth differences in male or female white sucker between the reference and exposure areas. That is, suckers have comparable sizes at age and weights at length (condition).

Northern Pike

Northern pike, like white sucker, also differ in their growth characteristics by sex, with males being generally smaller than females. ANCOVAs are provided for males and females in Appendix 7. Unlike white suckers, however, northern pike were substantially different in size at age between areas. Both males and females were substantially larger at age in the exposure area than in the reference. Fish condition (length-adjusted weight) was similar in both areas. Thus, a fish of a given length had a comparable weight in both reference and exposure areas, but a specimen from the exposure area would tend to be younger. Reasons for the enhanced growth in pike in the exposure area are unclear, but appear inconsistent with a mining impact or possible reduced food base.

5.2.8 H8 - Organ Size as a Response to Exposure

Summaries of data on liver weight, gonad weight and fecundity (numbers of eggs) are presented along with ANOVAs in Appendix 7 for both northern pike and white sucker. Males and females are treated separately for all measurements, and results are adjusted for fish age or fish size.

White Sucker

In white sucker, both body weight and age-adjusted liver weight in males showed significant reference-exposure differences. In this case, mean liver weight at age and at body weight was greater in exposed fish than in reference fish. This could potentially be construed as an increase in liver energy storage, which may be inconsistent with a mine effect. The opposite response for gonad weight (slightly and statistically smaller gonads) occurred in exposed males, when the data were adjusted for body weight.

Northern Pike

In northern pike, all three parameters showed significant reference-exposure differences. Liver weight, gonad weight (both sexes) and fecundity were greater in Bell Creek northern pike (applies

to age and/or body-weight adjusted data). These findings are consistent with the greater size of male and female northern pike at age - that is, larger fish have larger organs. However, the fact that females retain these organ size differences after body weight adjustment indicates that the responses are relatively strong. Overall, these responses appear to indicate more robust fish in Bell Creek. This may not be consistent with an obvious mine effect.

5.2.9 H9 - Dissolved vs Total Metal in Water as a Predictor of Biological Response in Fish

Hypotheses 9 through 12 involve examination of correlation coefficients between measured parameters. The correlations for H9 were computed using all reference and exposure area CPUE, BPUE, fish growth and organ size/fecundity measurements from Hypotheses H5 to H8 with emphasis on metals that showed significant area differences. The correlation matrix is shown in Appendix 7. The most relevant correlations are those for dependent variables (responses) showing reference-exposure differences consistent with a mining effect. However, as discussed for H5 to H8, it is suspected that the fish "effects" observed (increased growth and organ size in some cases) may be unrelated to metal exposure. Only the decreased body-weight adjusted organ size in female pike and male sucker appear consistent with a metal exposure impact.

For the most part, correlation coefficients were similar for dissolved and total metal fractions for some metals (Cu, Fe, Mg, Zn), but were higher for total lead than for dissolved lead. These data should, however, be interpreted very cautiously because, as noted above, most of these correlations may not reflect cause-effect relationships. Also, as noted previously, the mobility of fish within the localized exposure zone gradient and the inclusion of reference site data in the analysis limit the validity of any conclusions that can be drawn.

5.2.10 H10 - Metals in Sediment as Predictors of Biological Response

For H10 and H11, correlations were computed excluding the reference station data. As noted previously, it is appropriate to exclude the reference stations as the correlations more clearly reflect relationships within the mine exposure gradient, and are not potentially skewed by the extreme low values on the x-axis driven by the six reference area stations. Thus, a result producing a high correlation coefficient when tested with exposure station data only is more effective than one producing high values when reference station data are included.

For tool comparison, no statistical tests are performed to compare the correlations generated by two measurement tools. However, differences of 0.1 or more between coefficients are considered worthy of discussion, as long as at least one of the coefficients is statistically distinguishable from zero.

Tables showing correlation coefficients between sediment measurements (total, partial, SEM/AVS ratio) and benthic and sediment toxicity testing results are presented in Appendix 7.

Total and Partial Metals and SEM/AVS Ratios as Predictors of Benthic Community and Toxicity Response

Both total and partial metals were correlated with the benthic community responses identified in H6. Significant correlations were seen with many metals including As, Cd, Cu, Fe, Pb, Ni, Se and Zn. Neither the total nor the partial metal measurements were greatly superior in terms of the numbers of significant correlations or strength of the correlations, although total metals were slightly superior in terms of exhibiting the appropriate sign for the correlations and a slightly greater number of significant correlations. This observation is consistent with the fact that sediment chemistry trends were stronger for total metals. Both total and partial metals were more effective than the SEM/AVS ratio, which did not correlate with any benthic response.

Sediment toxicity correlations are evaluated using sediment toxicity endpoints, with a focus on *Chironomus* growth and *Tubifex* reproduction which showed significant among area differences in H1 (regardless of these differences being unrelated to exposure), as well as other sediment toxicity test results. As observed with benthic responses, significant correlations were found with both total and partial metals. Neither total nor partial metals was clearly superior as the independent variable group. The significant positive correlations between total metals and *Chironomus* growth appear unusual, but are consistent with the increased growth measured in sediment toxicity tests completed at Mattabi in the early 1990s (NTC and BEAK, 1993) and may indicate an unexplained but real effect of metal exposure in this case.

Overall, H10 results suggest that benthic community responses are slightly better correlated with total metals than with partial metals. This result is consistent with the observed exposure-related trends seen in H6 and in terms of the stronger spatial responses observed in total metals relative to partial metals. H10 results for sediment toxicity may be less conclusive in terms of distinguishing greater effectiveness in total or partial metals. However, if the positive correlations between total metals and *Chironomus* growth reflect a metal exposure response, then total metals could be

considered more effective than partial metals in this case. The SEM/AVS ratio was not correlated in a meaningful way with any benthic or sediment toxicity responses.

5.2.11 H11 - Sediment Toxicity as a Predictor of Benthic Community Response

Tables showing correlation coefficients between toxicity endpoints (*Chironomus* growth, *Tubifex* numbers of cocoons and young) and benthic indices (total density, numbers of taxa, % indicator taxa) are presented in Appendix 7. The toxicity parameters chosen for testing here include those showing significant among station differences in H1, even though these differences did not occur between reference and exposure areas. Other toxicity parameters were also considered including *Hyalella* results, and the correlation analysis broadened to consider dependent variables of interest that may not have been considered in H6. In particular, the analysis includes an examination of correlations between toxicity response in the laboratory and the abundance of the same or related organisms in the benthic community (e.g., is there a link between *Hyalella* survival and *Hyalella* density in the environment?).

Regardless of the lack of mining-related sediment toxicity effects in H1, several significant correlations occurred between sediment toxicity and benthic response. Correlations were strongest between the *Tubifex* reproduction endpoints and benthic endpoints. Thus, based on these comparisons, *Tubifex* reproduction responses appear to provide better indicators of benthic effects than *Chironomus* growth or other toxicity test results. It is interesting to observe the absence of correlations between *Hyalella, Chironomus* and *Tubifex* results and the relative abundances of these same taxa in the benthic community.

5.2.12 H12 - Total vs Dissolved Metals in Water as Predictors of Metal and Metallothionein in Fish

Tables showing correlation matrices between total and dissolved concentrations in water and fish tissue metal concentrations are presented in Appendix 7. Emphasis is placed on those metals and tissues showing significant mining exposure-related responses in H2 and H3.

Correlations could not be done for cobalt and selenium which showed significant exposure-reference area differences in tissue levels because aqueous selenium and cobalt were below detection limits in water at most stations. However, the possibility that cobalt and selenium concentrations in fish tissues could be affected by the mine cannot be discounted because,

although these metals showed little or no evidence of elevation in the exposure area, they were detected in the effluent.

The only significant responses in white sucker were for Pb and Zn in gill (both sexes) and for Pb in liver (females only). The correlation coefficients indicate a comparable linkage between metal in tissue and water for both total and dissolved forms. This conclusion is relatively weak, however, as it is based on a C/I design that precludes rigorous dose-response evaluation.

In northern pike, significant tissue-water correlations were found for MT in kidney and gill with aqueous Cd, Pb and Zn. Significant tissue-water correlations were found for Pb in kidney and gill, and Zn in muscle.

For most metals examined in this hypothesis, dissolved and total metals produced similar correlation coefficients. For cadmium, significant correlations could be identified only for total metal results, owing to the dissolved metals values falling below detection limits.

5.2.13 Triad Hypothesis

There are a number of chemistry (C), toxicity (T) and biology (B) monitoring tools that show significant bivariate correlations on all three arms of the "triad". For zinc, the correlations involving total metals were similar to those involving partial metals. However, for other metals, notably copper and cadmium, the correlations involving total metals were quite different than those involving partial metals. Spatial patterns of partial metals (except for zinc) did not follow those of total metals and were not clearly mine-related.

A more holistic evaluation of the sediment quality triad, involving multivariate analysis, is presented in Appendix 7. The many sediment chemistry variables were reduced by principal components analysis (PCA) to two sediment principal components (SPCs) representing sediment chemistry gradients. This PCA used both total metals and partial metals but not SEM/AVS results. Only partial and total metals appeared effective in hypothesis testing.

For partial metals, the dominant SPC1 (accounting for most (47%) of the overall variation in sediment quality) represents a mining-related gradient (Figure 5.8). Sediments influenced by mining tend to have a higher metal content, higher density, less moisture and less organic content than reference sediments. The subdominant SPC2 (accounting for 20% of the variation in sediment quality) represents variation in grain size, with more fine sediments at one end, versus sand at the other. The large lake stations (Mine Creek Bay and Peterson Cove) tend to contain more sand. Partial copper and cadmium tend to associate with sandy sediments rather than following the mining-related gradient in total metals.

For total metals, SPC1 accounts for 66% of the overall variation in sediment chemistry (Figure 5.9). SPC1 represents a mining-related gradient. As for partial metals, sediments influenced by mining tend to have a higher metal content, higher density, less moisture and less organic content than reference sediments. The dominant SPC2 represents mainly a grain size gradient, and explains only 14% of the overall sediment quality variation. It primarily separates Station MMR2-2 from all the others, based on fine texture. However, other reference stations and Mine Creek Bay stations tend to have more sand than No Name Lake stations.

The many benthic community variables were reduced by PCA to two benthic BPCs representing gradients in the biological make-up of the community. The dominant BPC1, accounting for 29% of the overall variation in species composition, primarily represents Hydracarina, Pisidium, Tanytarsini chironomids, ostracods and *Hyalella* (pollution sensitive taxa) as well as the chironomid *Psectrocladius* (pollution tolerant taxa) (Figure 5.10). Tolerant species dominate at No Name Lake, while sensitive species characterize the reference stations. The subdominant BPC2, accounting for 17% of the variation in species composition, represents *Chironomus* primarily, and separates Mine Creek Bay stations from all others.

The dominant partial metal gradient (SPC1) was significantly correlated with BPC1 (multiple $R = 0.81$, $p < 0.001$; Figure 5.11). This mine-related gradient (SPC1) was not significantly correlated with sediment toxicity, although SPC2 for partial metals was significantly correlated with *Chironomus* growth and *Tubifex* production of cocoons and young (multiple $R = 0.66$, $p = 0.019$). The large lake sediments (e.g., Mine Creek Bay and Peterson Cove), which group together on SPC2, tend to provide for better *Tubifex* reproduction than either

No Name Lake or Tag Lake sediments. However, "toxicity" does not appear to be driven by mine-related sediment features.

The dominant benthic community gradient, BPC1, representing sensitive vs tolerant species, was not significantly correlated with sediment toxicity, but toxicity was significantly correlated with BPC2, representing Mine Creek Bay species (i.e., better growth in Mine Creek Bay). The dominant benthic community gradient, BPC1, was related to the dominant sediment quality gradient (SPC1) which represents mine influence. Thus, the sediment features that drive toxicity (growth of test species) are different from those that drive benthic community composition. As noted in hypothesis testing (H1), relatively little toxicity was measured at Mattabi.

Based on Bartlett's sphericity test, and using only the dominant (mine-related) sediment quality and benthic community gradients, the sediment quality triad overall was significant. This demonstrates that chemistry, benthic and toxicity tools are linked, despite the weakness of the benthos-toxicity and chemistry-toxicity arms of the triad.

When the triad analysis was repeated using total rather than partial metals, the sediment-benthos arm of the triad was weakened and the other arms were not substantially altered relative to the partial metal triad (Figure 5.12). Results of this analysis are provided in Appendix 7. The sediment quality triad with total metals overall was significant despite the fact that none of the arms using the major PC axes were significant.

To illustrate an alternate approach, Mantel's test was performed in parallel with the previous analysis. Results are illustrated in Figure 5.13. For each of the benthic community, sediment chemistry (total metals) and sediment toxicity datasets (appropriately transformed), euclidean distance matrices were derived indicating overall similarities between pairs of stations. Concordance between the three pairs of distance matrices (benthos-chemistry, benthos-toxicity and chemistry-toxicity) was tested using Mantel's test (10,000 iterations). Overall, results indicated that none of the associations across each of the arms of the triad (B-C, B-T, C-T) was statistically significant, and the Bartlett's Sphericity Test was not statistically significant ($p > 0.05$). This solution supports the conclusion that the triad is relatively weak at Mattabi.

6.0 EVALUATION OF AQUATIC EFFECTS TECHNOLOGIES

6.1 Introduction

The Mattabi Field Evaluation program evaluated several of the aquatic effects monitoring "tools" considered by the AETE program. These tools were evaluated through testing twelve of the thirteen hypotheses pertinent to the 1997 field program, as well as by examination of other tool performance indicators other than those specific to these hypotheses (e.g., sediment quality triad, chironomid deformities, other cause-effect relationships, practical aspects). To avoid repetition, the cost-effectiveness aspects of the monitoring technologies are considered collectively in a summary report on all four of the 1997 field sites, because costs for each specific technology were approximately equal at the four sites (BEAK and GOLDER, 1998b). The summary report also evaluates the overall effectiveness of each monitoring tool, based on the results of all four sites.

Monitoring tools may be organized within "tool boxes" under the four guiding questions formulated under the AETE program to develop the hypotheses tested (from Section 1.1):

1. Are contaminants getting into the system?
2. Are contaminants bioavailable?
3. Is there a measurable (biological) response? and
4. Are contaminants causing the response?

Tool boxes and monitoring tools may be categorized under these four questions. Some tools may logically fit under more than one question; for example, toxicity testing tools may fit under Questions 1, 2 or 3. Table 6.1 provides a reasonable framework for organization of these tools, although alternate frameworks may be equally valid.

The fourth question cannot be answered by the application of individual tools, unlike the first three questions. Rather, the fourth question can be answered only by integrating the use of tools between and among tool boxes through testing for statistical linkages between potential cause and effect variables (e.g., do chemical concentrations and biological measurements correlate with one another?). The most effective tools are clearly those used in combinations that provide a yes answer to Question No. 4. The sediment quality triad represents a means of addressing Question No. 4.

The hypotheses are formulated to answer two general types of questions:

- Is the tool effective in measuring a mine effect (i.e., is there a reference - exposure difference or an exposure area gradient)?; and
- Is one tool more effective than another in measuring an effect?

The "effectiveness" of monitoring tools as discussed herein is specific to the Mattabi dataset. Mattabi represents one of four mine sites considered in the AETE 1997 Field Program, and only one of dozens of mine sites across Canada. A tool that is found to be of little value at Mattabi for detecting mine effects may be very useful at other sites and vice versa. Therefore, the reader is cautioned not to assume that the conclusions drawn from the Mattabi data will necessarily be broadly valid at mines across Canada. As shown in the AETE 1997 Field Program Summary Report (BEAK and GOLDER, 1998b), monitoring tools can respond very differently from site to site. Also, the presence or absence of a particular mine-related effect may simply reflect exposure level or bioavailability at the site. In the latter case, the absence of an effect may simply indicate that the tool was suitable for showing no effect.

6.2 Are Contaminants Getting Into the System?

6.2.1 Water Chemistry Tool Box

Hypothesis Testing Aspects

At Mattabi, water chemistry sampling in Bell Creek showed that metals were "getting into the system". This was demonstrated by elevated downstream concentrations in total and dissolved concentrations of several metals (e.g., zinc, copper, lead, cadmium, aluminum), and with increased selenium and cobalt concentrations measured in the effluent. In No Name Lake and Mine Creek Bay, sediment concentrations of cadmium, copper, lead and zinc clearly demonstrated that metals were getting into the system.

In testing of Hypotheses H9, elevated aqueous metal concentrations measured in Bell Creek were associated with enhanced fish size and fecundity at age. However, the effects observed for the most part were not consistent with an adverse impact, and the correlations may be a result of the C/I design and the responses more related to natural differences between reference and exposure areas. A few correlations were also observed between metal concentrations in water and tissue response (H12), with these responses more consistent with an exposure effect than noted for H9.

Other Considerations

TABLE 6.1: GUIDING QUESTIONS, TOOL BOXES AND TOOLS CONSIDERED IN THE 1997 FIELD PROGRAM. TOOL BOXES AND TOOLS IN BOLD PRINT ARE SPECIFICALLY CONSIDERED AT MATTABI

Question	Tool Boxes	Tools
Are contaminants getting into the system?	**Water chemistry**	• **total metal concentrations** • **dissolved metal concentrations**
	Sediment chemistry	• **total metal concentrations** • **partial metal concentrations** • **acid volatile sulphide and sequentially extracted metals**
Are contaminants bioavailable?	**Fish tissues**	• **organ/tissue metal concentration** • **organ/tissue metallothionein concentration**
Is there a measurable response?	Effluent chronic toxicity[1]	• fathead minnow survival and growth test • *Ceriodaphnia dubia* (microcrustacean) survival and reproduction test • *Selenastrum capricornutum* (algae) growth test • *Lemna minor* (duckweed) growth test
	Sediment toxicity	• ***Chironomus riparius*** **(larval insect) survival and growth test** • ***Hyalella azteca*** **(crustacean) survival and growth test** • ***Tubifex tubifex*** **(aquatic worm) survival and reproduction test**
	Fish health indicators	• **fish growth (length, weight and age)** • **fish organ size, fecundity**
	Fish population/community health indicators	• **fish catch-per-unit-effort (CPUE - by species and total)** • **fish biomass-per-unit-effort (BPUE - by species and total)**
	Benthic community health indicators	• **densities of benthic invertebrates** • **numbers of benthic invertebrates** • **benthic community indices (e.g., EPT index)** • **frequency of chironomid deformity**
	Periphyton community health indicators	• periphyton community biomass • numbers of periphyton taxa
Are contaminants causing the response?	**Pair-wise combinations of the above tool boxes**	• **chemistry x biology tool correlations** • **toxicity x biology tool correlations** • **chemistry x toxicity tool correlations** • **Sediment Quality Triad**

[1] Effluent chronic toxicity measured in the laboratory may also be categorized under Questions 1 or 2 (Are contaminants getting into the system?,or, Are contaminants bioavailable?).

The collection of dissolved metal samples according to the methods described in Annex 1 (provided under a separate cover) was not onerous, but required approximately five technician hours (additional relative to total metal samples) to filter and preserve the 18 samples (15 plus field duplicates and effluent samples).

The syringe and filter apparatus required, based on recommendations by chemists with the Geological Survey of Canada (GSC), were difficult to procure in Canada. Importation of the syringes from the U.S. required over one month due to delays at Canada Customs; thus, syringes were borrowed from GSC until delivery of our order. Availability of similar filtration materials necessary for ultra-trace metal work may be problematic in the future, requiring careful planning.

The commercial laboratory used required very specific instruction to provide sampling containers and filtration materials consistent with the specifications provided by GSC. For example, commercial laboratories often provide low density rather than high density polyethylene containers for metal samples, and may also provide containers with coloured lids such as "Falcon" tubes to consultants or mining companies. GSC has shown that such containers can contribute low levels of metals to water samples, and thus may not be suitable in aquatic effects monitoring where metal concentrations of interest are equal to or often below surface water quality guidelines.

The filtration procedure involved squeezing the water through a syringe-mounted filter, and was somewhat difficult and time-consuming due to the slow rate of filtration, rinsing requirements, etc. Also, where suspended solids levels are higher, filters became quickly clogged and required replacement.

Sample contamination was generally not apparent in the dissolved metal results, as dissolved metal concentrations were generally less than or equal to total metal concentrations (with exceptions occurring mainly at low concentrations near the detection limits).

To conclude, water chemistry (metal concentration) measurements were effective predictors of exposure at Mattabi. Neither total nor dissolved metal concentrations were more effective predictors of impacts in fish.

6.2.2 Sediment Chemistry Tool Box

Hypothesis Testing Aspects

In the exposure areas of No Name Lake and Mine Creek Bay, sediment concentrations of most metals demonstrated that contaminants were getting into the system. The gradients were most strongly demonstrated for total metals. The sediment chemistry tools of total metals, partial metals and SEM/AVS were evaluated through Hypotheses H10 by identifying reference versus exposure differences or concentration trends within the exposure gradient, and by examination of sediment metal correlations with biological responses (both benthic indices and sediment toxicity).

In general, reference-exposure differences and exposure area trends were observed for Zn, Cd, Cu, Pb, Ni and to a lesser degree for Hg and As.

Total metal and partial metal concentrations provided value in predicting biological effects in benthic communities or sediment toxicity. Neither total nor partial metals showed substantially better correlations with benthic community responses or toxicity, although correlations with total metals tended to be slightly superior in terms of the overall numbers of significant correlations. The SEM/AVS results did not show any significant correlation with the benthic metrics. There was only one weak correlation with a sublethal sediment toxicity response, but in the wrong direction, indicating that this sediment tool was not effective in predicting effects at Mattabi.

Other Considerations

Total metals and partial metals were similar in effectiveness in predicting biological effects, although the overall sediment quality triad was stronger when calculated with partial metals. The use of partial metals requires that the field crew have access to a freezer or dry ice since the samples have to be frozen after collection. The samples must also be kept frozen during transport to the analytical laboratory. In some field situations, this could increase the cost of sample collection, further decreasing the cost-effectiveness of this tool.

Sediment metal analyses may be more effective than aqueous metal analyses situations where aqueous metal concentrations are affected only sporadically (e.g., only in response to runoff or to intermittent effluent discharge), with concentrations approaching natural background between these impact events. This is because sediments will act to integrate metal loadings gradually over time while the water column may flush more rapidly.

The ineffectiveness of AVS and SEM determinations is perhaps not surprising, given the underlying assumptions in the SEM/AVS model. The SEM/AVS model relates the molar concentration ratio of potentially toxic simultaneously extracted metals (Cd, Cu, Pb, Ni, Zn) to the

molar concentration of amorphous solid metal sulphide (predominantly FeS; Allen *et al.*, 1993). Where the SEM/AVS ratio is >1.0, some of the metals are not rendered unavailable by formation of metal sulphides and toxicity may occur (e.g., Long *et al.*, 1998). At lower ratio values, toxicity should not occur. In many mining-impacted sediments, including those in No Name Lake and Mine Creek Bay, metals are often introduced to the environment in complex metal sulphide minerals in tailings or other solids, and may not be controlled in their mobility by simple monosulphide forms. The potentially large fraction of sulphide mineral present and the uncertain behaviour of minerals such as pyrite (iron sulphide), sphaleride (zinc sulphide), chalcopyrite (copper sulphide) and galena (lead sulphide) in the extraction potentially introduces a major uncertainty relating to the assumptions associated with the SEM/AVS model.

6.3 Are Contaminants Bioavailable?

This question is answered through the measurement of metal bioaccumulation or biochemical responses to metal bioaccumulation (i.e., MT). Overall, the Mattabi results suggest that metals from the mine are sparingly bioavailable to fish in Bell Creek.

6.3.1 Tissue Metal Concentrations

Hypothesis Testing Aspects

The effectiveness of tissue metal concentrations as indicators of metal bioaccumulation is determined by the identification of differences between exposure and reference areas, with higher values in the exposure area indicating effectiveness. Tissues showing greater exposure-reference differences are considered more effective than those showing smaller differences for the same metal.

The relative affinities of metals for specific tissues tended to be the same in the two species. For example, copper accumulated principally in liver, cadmium and lead in kidney, and selenium in both liver and kidney in both species.

Reference-exposure area differences in tissue metal concentrations in sucker were evident only for Pb and Zn, and only in gill among the key metals associated with Mattabi (Zn, Cu, Pb, Cd). The degree of difference for Zn, in particular, is very small (<10% difference in area means; refer to Table 4.10). These results suggest that the principal metals from Mattabi are low in bioavailability (or are efficiently excreted) in white sucker. Selenium, however, was more concentrated in all tissues in exposed sucker (but not always in both sexes) and appeared to respond more effectively

than any other metals; however, the connection of a selenium source with the mine is somewhat tenuous.

As in white sucker, the principal metals from Mattabi showed limited bioavailability in northern pike, with slightly greater tissue concentrations evident in the exposure area for Cd (muscle), Pb (gill and kidney) and Zn (muscle). As noted for sucker, Se and Co also appeared to respond, although for both metals a source linkage with Mattabi is not in strong evidence. Thus, tissue metal concentrations were apparently responsive to exposure in Mattabi pike, but only weakly so for the key metals. In neither species was a response in tissue copper observed.

Hypothesis 12, which compares correlations between metals in water and metals in fish tissues, showed significant correlations for Pb in liver of female sucker, and for Pb and Zn variously in gill, kidney and muscle of sucker and/or pike. These correlations are consistent with exposure-reference differences in H2. Hypothesis 12, however, is probably less effective in testing tissue metal tools for Pb than is H2, because of the large number of non-detect lead and chromium concentrations in the water quality data set. Correlations of tissue concentrations of Co and Se with waterborne levels could not be determined in H12 because these metals were below detection limits at most sites.

Other Considerations

From a practical standpoint, collection of tissues for metal analysis was not problematic, although more effort was required for fish collection and dissection than was necessary for small fish viscera at other mine sites in the 1997 AETE Field Program. The cold water conditions in October 1997 were conducive to maintaining viable fish for dissection, although viability was necessary for MT rather than for metals.

The correlations observed between concentrations of some metals in tissues and MT in tissues were good in some cases, especially in northern pike liver for Cu and the molar sum of Cd + Zn + Cu. These correlations imply cause-effect relationships within the tissues. The correlations between tissue metals and tissue MT in both species, especially in liver, imply metals and MT in liver effectively respond to accumulated metals, although the accumulation is not clearly connected with a mine effect as indicated previously.

6.3.2 Tissue Metallothionein Concentrations

Hypothesis Testing Aspects

The effectiveness of tissue MT concentrations as indicators of exposure to bioavailable metals from mine exposure is determined by identification of differences between exposure and reference areas, with higher values in the exposure area indicating effectiveness. Where more than one tissue type (gill, kidney, liver) shows a significantly elevated exposure area response, the tissue(s) having larger exposure-reference differences are identified as more effective.

At Mattabi, significant MT responses were found in northern pike only, with responses observed in gill and kidney. No response was identified for pike liver. This limited response in MT at Mattabi appears consistent with the limited metal bioaccumulation measured in Mattabi fish.

Other Considerations

The collection of tissues for MT analysis was not problematic, although the effort required for sample collection was greater than for fish viscera for other 1997 AETE field sites. The cold water conditions of October 1997 were conducive to maintaining fish viability until dissection, as required for MT. Maintenance of a dry ice supply was logistically difficult and expensive, with the supply delivered every three days by bus from Winnipeg to Ignace, and by taxi from Ignace to field headquarters.

Liver MT levels were strongly correlated with liver metal levels for Cu and the molar sum of Cd + Cu + Zn in northern pike. Similar correlations were not seen for other tissues. This result implies a more effective MT response in liver than in other tissues once metal bioaccumulation occurs, although the metal source appears unrelated to Mattabi (i.e., liver Cd, Cu and Zn concentrations did not show a reference-exposure difference).

6.4 Is There A Measurable Effect?

The answer to this question is evaluated through Hypotheses H1, and H5 through H12. The hypotheses tested at Mattabi are based on a measurable effect in fish and benthos (H5 through H8) and on the integration of tools hypotheses (H9 through H11) which look for correlations between the measurable effect and the possible causal agents. Overall, the results suggest that the benthic community is affected by exposure, but that measurable sediment toxicity, as tested in H1, is not. Reasons for this paradox are unclear.

6.4.1 Sediment Toxicity

Hypothesis Testing Aspects

The effectiveness of sediment toxicity as an indicator of metal bioavailability is determined by the identification of differences in toxicity between reference and exposure areas and/or the occurrence of trends within the exposure areas (near-field to far-field). Effectiveness is also determined by the strength of correlations between possible causal agents (metals in sediment) and sediment toxicity and between sediment toxicity and the benthic community.

Sediment toxicity was evident in all three test species, including mortality and growth impairment in *Hyalella* and *Chironomus*, and reproductive impairment in *Tubifex*. However, none of these responses could be related to a mining effect in H1. Additionally, toxicity was not strongly correlated with sediment chemistry or associated with the SEM/AVS ratio. These results suggest that metals in exposure area sediments, although high in concentration, were low in bioavailability. Despite the general lack of strong linkages with sediment chemistry or with exposure to mine effects, some correlations were found between toxicity, especially *Tubifex* reproduction, and benthic response. These correlations may be spurious, as they are inconsistent with any toxicity-benthos linkage in the triad analysis, or with any other evidence. Overall, sediment toxicity generally responded poorly or not at all to sediment contamination at Mattabi.

These results underscore the importance of analyzing field reference sediments in evaluating sediment toxicity, because field references proved toxic in some cases.

Other Considerations

From a practical standpoint, sediment toxicity was readily assessed at Mattabi. *Hyalella* and *Chironomus* showed reduced survival in some sediments, while *Tubifex* showed no significant lethality response. *Tubifex* testing is not currently widely available from commercial laboratories. Commercial testing capability is widely available for sediment testing with *Chironomus* and *Hyalella*.

6.4.2 Benthic Community Health Indicators

Hypothesis Testing Aspects

Monitoring of benthic community parameters was effective in identifying response to mining effects in the exposure areas at Mattabi, with effects on total density, total numbers of taxa and on specific sensitive and tolerant indicator species. This effectiveness was evident in terms of

reference-exposure differences and with respect to correlations with sediment metal concentrations in H10 and in the triad. No associations were seen between benthic indices and SEM/AVS results, suggesting that this is not an effective tool in predicting benthic effects at Mattabi.

Other Considerations

The collection of benthos for analysis at Mattabi was accomplished readily and required routine effort.

The incidence of chironomid deformity and abnormality, based on examination of mouth parts in mounted specimens, was low throughout the reference and exposure areas (Appendix 5), indicating that this tool would be ineffective in measuring biological responses to metals at Mattabi.

6.4.3 Fish Health Indicators

Hypothesis Testing Aspects

Fish health indicators, including community level indicators (catch/biomass-per-unit-effort, number of taxa) and population/individual level indicators (growth, organ size) were not rigorously tested at Mattabi owing to the limitations imposed by the C/I design. Nonetheless, most of the fish health indicator evidence shows little negative impact of mine exposure on fish abundance or biomass, and effects on numbers of taxa are likely related to reference-exposure area habitat differences. When reference-exposure differences in fish growth were evident (northern pike), the differences did not support a metal impact conclusion (i.e., exposed fish grow faster). Similarly, liver weight, gonad weight and fecundity responses, which occurred in both species, did not support an adverse mine impact conclusion (i.e., exposed fish tended to have larger livers and gonads when adjusted for age and/or body weight).

Overall, the general absence of "negative" impacts on fish health, such as reduced abundance, growth or reproductive performance, are consistent with the results of fish tissue analyses. That is, tissue analyses imply that key metals from Mattabi are low in bioavailability, as measured by determination of tissue metal or MT concentration; accordingly, little deleterious impact should be expected.

Other Considerations

Fish measurements were readily taken and dissections readily performed in the field. The skill sets used here are the same as required in support of the adult fish survey of environmental effects monitoring programs for the pulp and paper industry.

Given that significant differences occurred in fish health indicators between the Mattabi reference and exposure areas, and that the differences do not support a mine impact conclusion, this indicates some important aspects with respect to the interpretation of these data. If the differences observed were exposure-related, then exposure to metals at non-toxic concentrations can potentially produce more robust fish. If the differences are unrelated to metal exposure and are simply in response to the sampling of distinct fish populations, then the probability is high at other sites of declaring a mine impact when the causes of the response differences are more likely associated with other (natural) factors. The probability is particularly high within a C/I study design framework such as implemented for fish at Mattabi.

6.5 Are Contaminants Causing the Responses?

As indicated previously, this question is not answered directly through the application of specific monitoring tools evaluated in this study, or through any of the hypotheses tested. Rather, the question is evaluated only by a weight-of-evidence provided by affirmative responses to the first three questions, and particularly by the strength of correlations between exposure indicators (chemical concentrations) and biological responses in hypotheses H9 through H12. Strong linkages between sediment chemistry, sediment toxicity and benthic community characteristics in the sediment quality triad would provide further support that contaminants cause the responses.

At Mattabi, evidence indicates that contaminants (metals) are getting into the system. However, analyses of fish tissues for metals and MT suggest that metals in Bell Creek are sparingly bioavailable. The biological responses observed in fish (growth, reproductive indicators, organ size) are generally not indicative of an adverse effect, a result that is consistent with low metal bioavailability.

If sediment toxicity can be considered to reflect bioavailability of metals in sediment, sediment metals at Mattabi are low in bioavailability (i.e., sediment toxicity was not strongly related to metal concentrations). This absence of strong metal-related toxicity is unusual, given the very high concentrations of metals such as zinc in the sediments.

Benthic community responses are evident and are associated with proximity to mine sources and with metal concentrations in sediments. This result implies that benthic responses are associated with contaminants and that benthic community responses can be more sensitive than sediment toxicity responses.

The sediment quality triad produced either weak or insignificant linkages between sediment chemistry, sediment toxicity and sediment biota, depending on the statistical solution and datasets used. The strongest linkage was found between partial metal concentrations and benthos. These results provide limited support for cause-effect relationships between chemistry and biota, chemistry and toxicity, and toxicity and biota. One may conclude that sediment metals are relatively low in bioavailability, to such an extent that other factors (e.g., minor grain size differences, natural "toxicants", etc.) tend to mask any impacts of sediment-associated metals.

Based on the above considerations, the Mattabi example serves to illustrate that metals released to the environment from mining activities can in some cases produce little impact, despite the presence of high concentrations in sediments and water. Hypotheses tested here also show that tools thought to better measure bioavailable metals (e.g., dissolved metals in water, partial metals in sediment) are not necessarily more effective than total water and sediment analyses.

6.6 Section Summary

Table 6.2 provides a summary of the effectiveness rankings of the aquatic monitoring tools evaluated at Mattabi. Table 6.3 compares the effectiveness of alternate tools that may be used to measure metal concentrations, metal bioavailability or biological response.

Some of the tools evaluated demonstrated a mine effect at Mattabi whereas others did not. Tools demonstrating or partially demonstrating effects at Mattabi included most in the water and sediment chemistry tool boxes (with the exception of SEM/AVS), most in the benthic community tool box, and some of the fish tissue and MT tools (i.e., all tissues were effective, but for few metals and with differences between species and sexes). Tools that did not demonstrate effects included several in the fish health, fish population/community, and

sediment toxicity tool boxes. The limited effectiveness of some of these tools may be due to low metal bioavailability, possibly combined with small differences in habitat, food abundance, etc. Of the tools in the same tool box ranked as demonstrating effects, major differences in effectiveness were not evident at Mattabi. This indicates that cost-effectiveness will be important in evaluation of these tools. Cost-effectiveness is assessed in a separate summary report on the four 1997 field sites (Heath Steele, Myra Falls, Dome and Mattabi) (BEAK and GOLDER, 1998a).

A tool was considered to partially demonstrate an effect if it occurred with a limited number of endpoints, if the response was minimal, or if the response was in a direction inconsistent with adverse impact. For example, total and dissolved metals in water showed a partial effect in that they were unrelated to any adverse health impacts in fish, but did correspond with some fish tissue metal responses. Fish tissue metal concentrations were affected but in many cases not in terms of the key metals from the mine; thus, the effects were partially demonstrated. Liver weights adjusted for age were greater in exposed sucker and pike – an effect not clearly consistent with adverse impact and therefore considered a partial effect.

Overall, the relatively subdued impacts of metals in the environment near Mattabi are unexpected and noteworthy. This result contrasts with the greater bioavailability and impact observed at the two other base metal mines studied in this program (Heath Steele, Myra Falls). This is particularly unusual considering that metal concentrations in sediments (e.g., Zn) at Mattabi are at higher concentrations than measured at other locations.

TABLE 6.2: EFFECTIVENESS OF MONITORING TOOLS TESTED AT MATTABI

Tool Boxes	Tools	Effectiveness			Comment
		Effect Demonstrated	Effect Partially Demonstrated	Effect Not Demonstrated	
Water Chemistry	Total Metals		√		Increased concentrations of Zn, Cu, Pb and Cd in exposure area. Se and Co source implied by effluent sample analysis. Increased concentrations appeared unrelated to any adverse effects on fish growth, community structure or organ size, but did show some relationships with tissue metal levels in fish.
	Dissolved Metals		√		
Sediment Chemistry	Total Metals Partial Metals	√	√		Gradients in exposure area evident, particularly for total Zn, Cu, Cd and Pb, but were weaker for partial metals. The corresponding correlations between partial metals and benthic responses were slightly weaker than those for total metals. Correlations occurred between sediment metals and the benthic community.
	SEM/AVS			√	SEM/AVS was not correlated to biological impact or toxicity. Chemical determination may be confused by the abundance of complex metal sulphide minerals in sediments.
Sediment Toxicity	*Hyalella azteca, Chironomus riparius, Tubifex tubifex*			√	No response to mine-related effects. Mortality and growth responses occurred in *Hyalella* and *Chironomus*, and reproductive responses occurred in *Tubifex*, although effects were unrelated to Mattabi
Fish Tissues	**White Sucker**				
	Metals: • Muscle		√		Unresponsive to mine exposure, except for Se. Evidence for a Se source in effluent is weak.
	• Liver		√		Unresponsive to mine exposure except for Se in females; weak correlations between liver MT and liver Zn, liver MT and liver Cu+Zu+Cu.
	• Gill	√			Responsive with respect to Pb, Zn and molar sum of Cd+Cu+Zn. No positive gill metal correlations with gill MT were found.
	• Kidney		√		Unresponsive to mine exposure, except for Se. No positive kidney metal correlations with kidney MT were found.

TABLE 6.2: EFFECTIVENESS OF MONITORING TOOLS TESTED AT MATTABI (cont'd)

Tool Boxes	Tools	Effectiveness			Comment
		Effect Demonstrated	Effect Partially Demonstrated	Effect Not Demonstrated	
Fish Tissues (cont'd)	**Northern Pike**				
	Metals:				
	• Muscle	√			Responsive for Cd, Zn, Se (both sexes). Evidence for a Se source at mine is weak.
	• Liver		√		Unresponsive to mine exposure except for Se. Liver MT strongly correlated with liver metals for Cu and Cd+Cu+Zn.
	• Gill	√			Responsive for Pb, Co and Se. Meaningful correlations between liver MT and liver metals were not evident.
	• Kidney	√			Responsive for Pb, Co (both sexes) and Se (females only). Evidence for Se and Co source at mine is weak. Weak correlation between kidney MT and kidney lead.
	White Sucker MT				
	• Liver			√	Liver MT unresponsive to mine exposure. Correlated with hepatic Zn and Cu+Zn+Cd.
	• Gill			√	Gill and kidney unresponsive to mine exposure. No positive correlations between gill/kidney MT and gill/kidney metals.
	• Kidney			√	
	Northern Pike MT				
	• Liver			√	Hepatic MT unresponsive to exposure. Strong correlations between hepatic metals (Cu, Cd+Cu+Zn) and MT.
	• Gill	√			Gill and kidney responsive to mine exposure. Positive correlations between gill/kidney MT and tissue metals were absent or weak.
	• Kidney	√			

TABLE 6.2: EFFECTIVENESS OF MONITORING TOOLS TESTED AT MATTABI (cont'd)

Tool Boxes	Tools	Effectiveness			Comment
		Effect Demonstrated	Effect Partially Demonstrated	Effect Not Demonstrated	
Fish Health Indicators	**Body Size and Age (Growth)**				
	• White sucker			√	No difference in weight at age for reference and exposure fish.
	• Northern pike			√	Exposed fish not negatively affected, but much larger at age. Fish condition (weight at length) unaffected.
	Liver Weight				
	• White sucker		√		Liver weight (adjusted for age or body weight) greater for male exposure fish. Nature of "effect" not clearly mine-related.
	• Northern pike		√		Larger liver weights (at age or body weight) in exposed females, and males (at age only). Nature of effect not clearly mine-related.
	Gonad Weight				
	• White sucker		√		Body weight-adjusted gonad weight slightly reduced in exposed males (potential mine-related effect).
	• Northern pike			√	Larger gonad weights for exposed fish. Effects not clearly mine-related.
	Fecundity				
	• White sucker			√	No difference in fecundity at age or body weight.
	• Northern pike			√	Greater fecundity at age but not at body weight. Effects not clearly mine-related.
Fish Population/ Community Health Indicators	**CPUE**				
	• White sucker			√	CPUE was slightly (but not significantly) greater in exposure area. Effect not clearly related to exposure.
	• Northern pike			√	CPUE for pike was comparable in reference and exposure areas.

	• All fish species			√	CPUE for all fish was comparable in reference and exposure areas.

TABLE 6.2: EFFECTIVENESS OF MONITORING TOOLS TESTED AT MATTABI (cont'd)

Tool Boxes	Tools	Effectiveness			Comment
		Effect Demonstrated	Effect Partially Demonstrated	Effect Not Demonstrated	
	BPUE				
	• White sucker			√	BPUE was comparable in reference and exposure areas for sentinel species and all fish.
	• Northern pike			√	
	• All fish species			√	
	Number of species			√	Numbers of species in catches were greater in reference area than exposure, owing to presence of coregonids exclusively in the reference. This is believed to be due to a habitat effect unrelated to exposure.
Benthic Community Health Indicators	Benthic Density	√			Exposure-reference difference and exposure area trend observed.
	No. of Taxa		√		Exposure area trend observed, but no reference-exposure area difference.
	Abundances of Indicator Species	√			Reference-exposure differences and exposure area trends evident in *Chironomus, Pisidium* and Hydracarina abundances.

TABLE 6.3: COMPARATIVE EFFECTIVENESS OF MONITORING TOOLS AT MATTABI

Tools	Comparison
Total Metals vs Dissolved Metals in Water	Total and dissolved metal concentrations approximately equal in reflecting elevated metal concentrations. Concentrations of both appeared unrelated to most biological effects, although some correlations occurred between metal concentrations and tissue response.
Total Metals, Partial Metals and SEM/AVS in Sediment	Total and partial metals were, on average, similar in reflecting benthic effects, with total metals slightly better correlated than partial metals. Total metals also better reflected mine gradients (showing "contaminants getting into the system"). The SEM/AVS ratio was unrelated to benthic effects or sediment toxicity.
Sediment Toxicity Tests	None of the tests indicated mine-related impact (i.e., reference-exposure differences or exposure area gradients). *Tubifex* was not effective as a survival test. Absence of mine-related sediment toxicity precluded more rigorous evaluation of tests.
Benthic Community Health Indicators (density, no. of taxa, indicator taxa)	Several indices indicated mine-related impact including total density, no. of taxa, and abundance of Hydracarina, *Chironomus* and *Pisidium*. Of these indicators, number of taxa and % Hydracarina were most strongly correlated with sediment metal concentrations.
Fish Tissues - Metals	In sucker and pike, no tissue was clearly superior in responding to mine exposure overall. In sucker, gill was the only tissue responding to any of the major Mattabi metals. In pike, gill, kidney and muscle all responded in terms of some of the major metals. Liver responded effectively to bioaccumulated Cu+Cd+Zn and Cu in pike, but metal source in liver unrelated to mine exposure.
Fish Tissues - Metallothionein	In sucker, MT levels did not respond to mine exposure in any tissues. In pike, gill and kidney were responsive to exposure. Pike liver from all reference-exposure fish responded to accumulated metals (Cu, Cd+Cu+Zn) but response unrelated to mine exposure.
Fish Tissues - Metals vs Metallothionein	In the only tissues where MT and metals both responded to exposure (gill and kidney in pike), MT and tissue metals responded similarly. For MT and tissue metals overall, effects were more often demonstrated for metals than for MT.
Fish Health Indicators	Among the responses observed (growth, condition, liver weight, gonad weight, fecundity), most were inconsistent with adverse mine effects. A small reduction in gonad size in exposed male sucker could be construed as a mine effect. Liver weight increases in both male sucker and in pike, gonad weight increases in pike and increased somatic growth in pike are all less obviously related to mine exposure.
Fish Population/Community Health Indicators	Neither CPUE, BPUE nor numbers of fish taxa were responsive to mine exposure.

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